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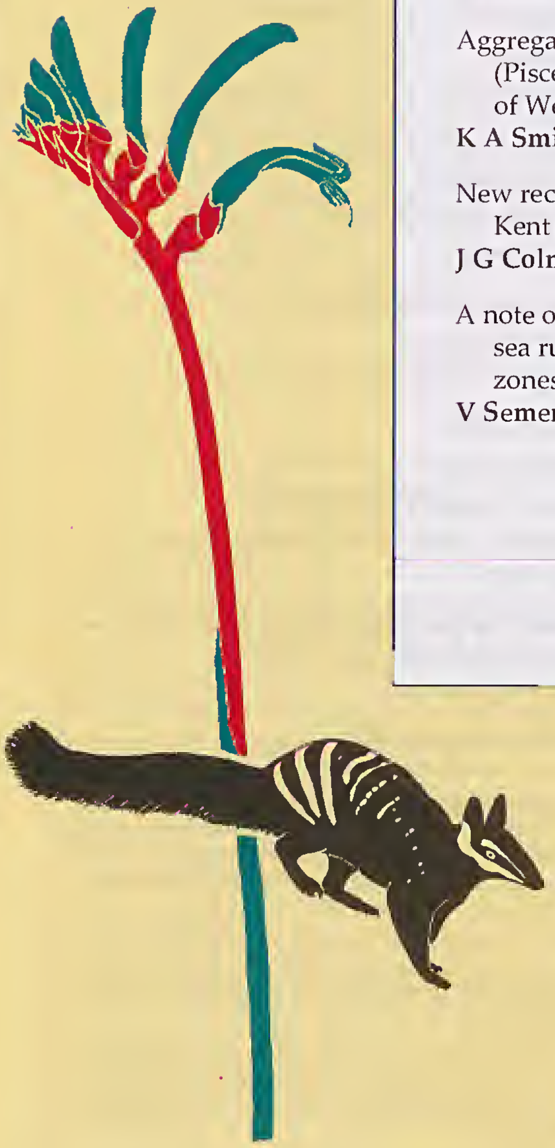

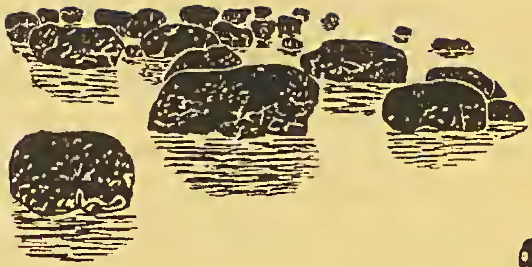

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Cover design: The four subjects symbolize the diversity of sciences embraced by the Royal Society of Western Australia. Mangles' kangaroo paw (*Anigozanthos manglesii*) and the numbat (*Myrmecobius fasciatus*) are the floral and faunal emblems of Western Australia, and stromatolites are of particular significance in Western Australian geology (artwork: Dr Jan Taylor). The Gogo Fish (*Mcnamaraspis kaprios*) is the fossil emblem of Western Australia (artwork: Danielle West after an original by John Long).

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Assessing the short-term effect of minerals exploration drilling on colonies of bats of conservation significance: a case study near Marble Bar, Western Australia

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Abstract

Bats are most vulnerable whilst in their roost, and activities that result in roost destruction or disturbance have the potential to cause declines in species of conservation significance. However, conservation efforts for bat colonies can be limited by a lack of understanding of the effect of certain disturbances. An evaluation drilling programme conducted in close proximity to historical underground gold workings near Marble Bar provided an opportunity to examine the short-term effect of this type of activity on colonies of the bats *Macroderma gigas* and *Rhinonicteris aurantia*. A non-invasive approach to assessing the impact of the associated activity was developed, which simultaneously realised the best economy of moving a drill rig. Bats were subject to several types of potential disturbance (from noise and vibration) from earthmoving equipment, the drill rig and the booster compressor. Monitoring involved continuous acoustic and visual observations of mine entrances during drilling, direct counts of emerging bats each evening after drilling, and surveys of other mines in the local area throughout the study. *R. aurantia* was present throughout the drilling programme, but actual numbers could not be determined accurately. A marked increase in the number of *M. gigas* was observed, thought to be independent of the activities associated with the drilling programme and possibly due to concurrent human activities in other local mines or natural factors. It was concluded that short-term activities associated with drilling that are not within 25 m of a mine entrance or 85 m of the roost site within the structure, and that do not intersect workings, might not constitute a significant disturbance to these bats. However, the long term cumulative effects of this and other disturbances, and in the context of contemporaneous activities at other roosts in the district, are unknown.

Keywords: bats; conservation significance; Marble Bar; minerals exploration, drilling; roost destruction

Introduction

The diurnal refuge, or roost, of bats is a major influence in their ecology, offering protection from exposure to daytime ambient conditions and predators, promoting energy conservation, and facilitating social interactions (Kunz & Lumsden 2003). Bats are vulnerable to the destruction of their roosts, as well as disturbances while in them, which are issues for many species worldwide (Hutson *et al.* 2001; Mickleburgh *et al.* 2002). Cave roosting bats will readily move into disused underground mines and form colonies ranging in size from a few individuals to aggregations of thousands (Tuttle & Taylor 1998). The importance of mines to bats is exemplified by past actions in the USA, where the closure or destruction of thousands of mines led to the estimated loss of millions of bats before they were considered as habitat by land managers (Ducummon 2001; Altenbach & Sherwin 2002). In Australia, disused mines provide alternative habitat for more than 20 species of cave roosting bat, some of which are listed in a Threatened category by Australian State and Commonwealth authorities (Hall *et al.* 1997; QPWS 1999; ABS 2000; NSW National Parks and Wildlife Service 2001; Mickleburgh *et al.* 2002; Thomson 2002). The creation of underground mines has allowed some species to add breeding sites and expand their population size. The corollary is that now some mines are critical to the survival of numerous species because most of the known population is aggregated within them, (Duncan *et al.* 1999; Armstrong & Anstee 2000; Armstrong 2001; DEWHA 2010a).

The Pilbara, one of Australia's most heavily exploited mining provinces, contains isolated populations of two bat species of conservation significance that occupy caves and underground mines, and whose presence often coincides with the interests of gold, copper and iron ore mining. The Pilbara leaf-nosed bat *Rhinonicteris aurantia* (Hipposideridae) is listed as Schedule 1 under Western Australia (WA) State legislation, and as Vulnerable under Commonwealth legislation. The ghost bat *Macroderma gigas* (Megadermatidae) is listed on the WA Department of Environment and Conservation's (DEC) Priority Fauna List under "taxa in need of monitoring". Both species are widespread in the Pilbara region, however the majority of their number is concentrated in relatively few underground structures that meet their physiological and breeding requirements (Hall *et al.* 1997; Armstrong & Anstee 2000; Armstrong 2001; DEWHA 2010a). Most of these structures are disused mines that are still subject to interest for their undeveloped minerals potential (McKenzie *et al.* 1999; DEWHA 2010a). Both *M. gigas* and *R. aurantia* are also particularly sensitive to disturbance, and have an observed tendency to vacate roosts in response (Armstrong 2001; K.N. Armstrong unpub. obs.).

In the past decade, the mining boom experienced by WA has expanded our knowledge of their area of occupancy in the Pilbara through increased surveys as part of environmental assessments (K.N. Armstrong unpublished data), as well as baseline surveys conducted by the DEC (McKenzie & Bullen 2009). When a new record of presence occurs in an area planned for development, the proponent then has an obligation to address this as an issue for their project. In general, the success of conservation and management efforts for Threatened-listed species is dependent not only on a comprehensive understanding of their taxonomy and ecology, but also empirical knowledge of how they react to human disturbances and habitat modification. It might be possible to make predictions about such reactions based on knowledge of a species' ecology, but often the questions most relevant for development proposals are very specific and require past experience to answer. There are studies that have investigated how bats respond to disturbance following human entry of roosts, and also to solutions such as the placement of 'bat friendly' gates across cave and mine entrances (Johnson *et al.* 1991; Speakman *et al.* 1991; Richter *et al.* 1993; Thomas 1995; Lackie 2000, Mann *et al.* 2002; Pugh & Altringham 2005; Slade & Law 2008), but in the case of Pilbara mining projects the most common issues that arise when these species are discovered in areas planned for development do not have much similar published experience to draw upon. If plans do not involve the actual destruction of roosts, then a common consideration is the minimum proximity of mining infrastructure, pits and waste dumps to roosts before the combined impacts from noise, blasting, lights, and vehicular traffic cause the colonies to decline in size or reduce the frequency with which roosts are occupied (K.N. Armstrong unpubl.).

Before mining actually begins, the effects of exploration drilling in close proximity to caves or mines occupied by the bats sometimes needs to be considered. Making an assessment of effects such as noise and vibration on bats of conservation significance is difficult because there is no prior published information, and such applied studies require the cooperation and resources of mining companies to complete. An evaluation drilling programme (where the size and value of known ore bodies is determined by intensive drilling) conducted in the vicinity of the historical underground gold workings of the Klondyke Queen near Marble Bar provided the opportunity to examine the short term influence of this type of activity on *M. gigas* and *R. aurantia*. The two species have established colonies of regional importance in the workings, and the mine represents one of the few known roost sites for *R. aurantia* in the region (Churchill *et al.* 1988; Hall *et al.* 1997; Armstrong & Anstee 2000; Armstrong 2001; DEWHA 2010a). The evaluation drilling programme at the Klondyke Queen was used as a case study to determine whether such activities in close proximity to roosts would have a detectable effect on bat presence and abundance. The study was structured to examine bat presence before and during drilling, while simultaneously 1. introducing no confounding factor through the methods used to assess presence, colony size and nightly exodus; and 2. introducing no significantly disruptive or costly measure to the drilling programme.

Methods

Study species

The bats *M. gigas* and *R. aurantia* have similar distributions over northern Australia, including the Kimberley region, the Top End of the Northern Territory, and isolates in Queensland and the Pilbara region of Western Australia (Churchill 2008). They both rely on relatively deep subterranean roosts with a warm, humid microclimate that enable them to limit energy and water loss (Leitner & Nelson 1967; Kulzer *et al.* 1970; Churchill *et al.* 1988; Churchill 1991; Baudinette *et al.* 2000; Armstrong 2000, 2001), and have established strongholds in the eastern Pilbara where most underground mines occur. The majority of the current known population of *M. gigas* is concentrated in the Comet mine (Marble Bar), Klondyke Queen mine (Marble Bar), Lalla Rookh mine (Panorama Station), Bulletin and Bamboo Creek mines (Bamboo Creek), with smaller colonies (30 individuals or less, and often single individuals) in small mine adits and natural caves throughout the region. *R. aurantia* is found in fewer mines in the eastern Pilbara (Klondyke Queen, Comet, Bamboo Creek, Lalla Rookh, Copper Hills), but there have been numerous records of bats in flight throughout the region, and a record of a cave roost in Barlee Range Nature Reserve (Hall *et al.* 1997; Armstrong & Anstee 2000; Armstrong 2001; McKenzie & Bullen 2009; DEWHA 2010a; K.N. Armstrong, unpubl. data).

Human intrusion into roosting chambers and capture activities have been associated with reduced occupancy of these species at several caves and mines, so non-invasive methods are recommended for determining their presence (DEWHA 2010b). This usually requires electronic bat detectors that can record ultrasonic echolocation calls, but *M. gigas* can also be counted easily under low or red light when emerging from a roost because of their large size and pale colour. It is virtually impossible to make an accurate determination of colony size of *R. aurantia* using remote detection methods because they visit caves and mines after dusk that are not used for roosting during the day, and they will often swarm around a roost entrance after emerging at dusk (K.N. Armstrong, unpubl. data). If capture is not possible, then only presence and activity levels of *R. aurantia* can be determined at caves or mine entrances.

Two other species of bat commonly occupy underground mines in the eastern Pilbara: the common sheath-tailed bat *Taphozous georgianus* and Finlayson's cave bat *Vespadelus finlaysoni*. They are obligate cave roosting bats, and are typically abundant. Their responses to drilling disturbance were not considered in the study.

Study sites and timing

The study area included two main sites: the Klondyke Queen and Klondyke King mining leases, located c. 20 km south-east of Marble Bar, Western Australia (Figure

1). Drilling began at a third lease called Kopcke's Reward, c. 1.9 km east of Klondyke Queen. The observations made on bats were undertaken at the historical workings of the Klondyke Queen mine, which is the only existing underground structure of significant depth and complexity on these leases. A newer adit beneath the old Klondyke Queen is sealed and not known to be occupied by bats of conservation significance. Other historical mine workings in the Marble Bar area where *R. aurantia* had been detected

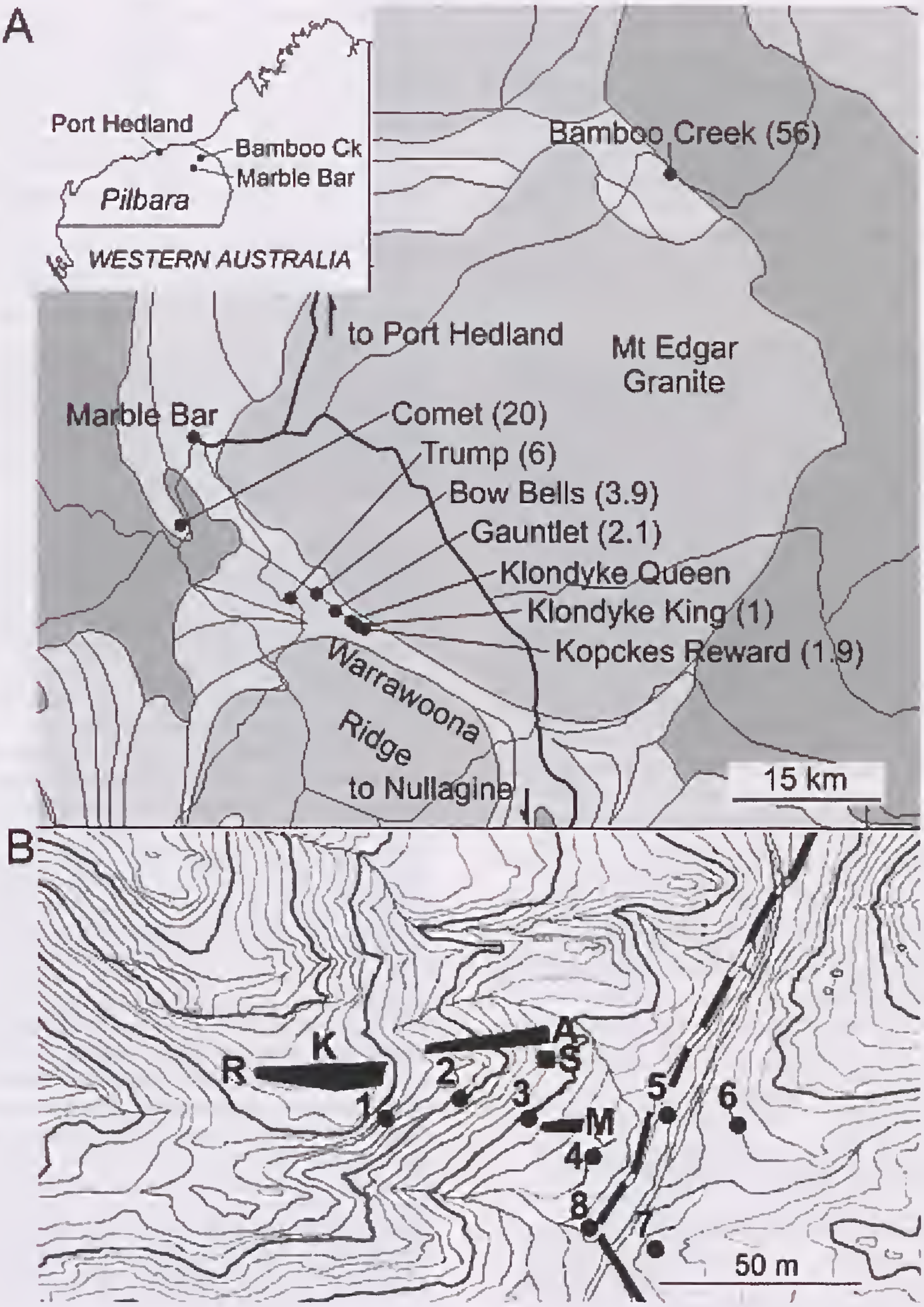

Figure 1. A: Inset: Location of Marble Bar in Western Australia. Main: Location of mines and leases examined for *R. aurantia* and *M. gigas* in the eastern Pilbara during the study, with the distance from the Klondyke Queen mine in parentheses. Shaded regions are major geological units (GSWA 2001). B: Location of drill holes relative to the underground workings in the Klondyke Queen project area (A: adit entrance to the Klondyke Queen; K: black polygons represent Klondyke Queen surface workings; M: entrance to the Mullan workings that extend below the older Klondyke workings; R: approximate location of roost site within the mine; S: small shaft adjacent the main adit entrance; broken line is a vehicle track).

previously (Armstrong 2001) were surveyed as part of the study. All activities were conducted between 12 June and 5 July 2001.

Study design

The study was designed to simultaneously minimise potential disturbance from the drilling programme and the investigators to avoid the exodus of *M. gigas* and *R. aurantia* from the Klondyke Queen mine, while at the same time minimising the cost of moving a drill rig and avoid leaving it idle. Moving a drill rig is costly since it increases the time taken to complete an drilling programme, and was a major inconvenience in the present study area given the steep terrain and low quality of drill pad access tracks. The type of drilling was Reverse Circulation (RC), which results in the extraction of fine, dry dust, and sometimes requires the use of a truck-mounted air compressor ('booster') to provide extra power for deeper holes, and which is relatively noisy compared to the drill rig.

The approach consisted of a logical order to the progression of drilling, and incorporated a set of measurements designed to detect a set of anticipated responses from the two bat species, and which were feasible to be implemented at the site. Observations were made on the bats before drill pad and access track construction with a bulldozer, between pad construction and the commencement of drilling, and throughout the drilling operations (Table 1). The magnitude of the potential disturbance was related to a combination of several factors, including the distance of the rig from the mine entrance, the time taken to drill (dependent on hole depth), whether the booster was used, and what time of the day a particular hole was commenced (early morning or midday). Each hole was then classified in terms of the anticipated level of disturbance, and a schedule was derived that alternated drilling between the Klondyke Queen (eight holes) and Klondyke King areas (12 holes; Table 1; Figure 1). The goal was to gradually increase the level of anticipated disturbance, while determining simultaneously whether an effect could be observed. The schedule could be adjusted based on these responses, and also contingency measures were prepared in case a significant and undesirable effect was observed during drilling.

Table 1

Activities undertaken during the study. Counts of emerging bats were made 17:45 – 20:00 on all nights except 18 – 19/6/01.

12/6/01	Pre-drill pad construction monitoring (14:30 – 17:45); Surveying nearby mines/leases (Kopcke's Reward, Klondyke King, Klondyke Boulder, The Gauntlet, Bow Bells, The Trump).
13/6/01	Surveying nearby mines (Comet, MacKinnons).
18/6/01	Drill pad construction with bulldozer (14:15 – 17:15) (author absent).
19/6/01	Drill pad construction with bulldozer (14:15 – 17:15) (author absent).
29/6/01	Full day drilling at Klondyke King project area; Pre-drill monitoring (14:30 – 17:45).
30/6/01	Morning drilling at Klondyke King project area; Hole 5 drilling (14:00 – 17:15): monitoring (13:20 – 17:45).
1/7/01	Morning drilling at Klondyke King project area; Hole 4 drilling (13:20 – 17:15): monitoring (12:40 – 17:45).
2/7/01	Morning drilling at Klondyke King project area; Hole 3 drilling (13:30 – 16:20): monitoring (13:20 – 17:45).
3/7/01	Hole 7 drilling (08:30 – 17:30): monitoring (07:15 – 17:45).
4/7/01	Hole 8 drilling (07:40 – 17:00): monitoring (07:25 – 17:45).
5/7/01	Hole 6 drilling (07:40 – 13:30): monitoring (07:40 – 17:45).

Drilling commencement time allowed for morning, afternoon, or full-day drilling in the Klondyke Queen area. Drilling was planned initially for mornings in the Klondyke King area (*i.e.* away from the roost in the Klondyke Queen workings) for several reasons: 1. it was predicted that there would be less impact on roosting *R. aurantia* in the afternoon when they become more active before their emergence, than in the morning when their circadian rhythms might dictate that they rest (although it is acknowledged that activity due to social interactions might occur throughout the day); 2. there was a lower likelihood of bats being exposed for the whole day if they vacated the mine due to drilling in the morning; 3. if a response to drilling was detected, the drill rig could be moved to the Klondyke King area, where work could be recommenced in the same afternoon to avoid having the rig idle; and 4. moving the rig to the Klondyke King for morning drilling might alleviate the cumulative effects of drilling a number of holes in succession close to the Klondyke Queen workings.

Drilling at the Klondyke Queen began in the afternoon at Hole 5 (Table 1; Figure 1), which was considered to have a low potential for disturbance given its shallow depth, relatively large distance from the mine entrance, and midday commencement time (Table 2). For the shorter holes, drilling for a slightly longer period with the rig only was considered to comprise less of a disturbance to bats than a shorter period with a greater surface noise level through the use of a booster. Therefore, the booster was not used on the shallow holes (Holes 5 and 3) closest to the Klondyke Queen workings, though its use was unavoidable on one hole of moderate depth (Hole 4) that was commenced at midday. Since there was no detectable effect of drilling the 'afternoon' Holes 5, 4 and 3, it was considered appropriate to commence the deeper Holes 7, 8 and 6 from early morning (the deeper holes required a full day and the use of the booster) (Table 2). The use of the booster was unavoidable on the deeper holes. It would have been expensive and unnecessary to drill the two deeper holes over two afternoons each since the drilling programme would have needed to be extended for a further two days. Holes 1 and 2 were not drilled and no drill pads were constructed for these. Pad construction would have involved blasting with explosives to batter down a steep slope adjacent to the Klondyke Queen workings close to bat roosts. Such activity was deemed highly likely to cause a disturbance to the bats and plans to drill these holes were therefore abandoned. The drill rig was always removed from the project area or shut down at or before 17:30 hours to avoid affecting bats emerging at dusk.

Measurements to detect anticipated responses of bats

Bats at the Klondyke Queen were potentially exposed to four types of disturbance from noise and vibration: associated with the bulldozer clearing access tracks and pads for the drill rigs; from the drill rig on the surface;

Table 2

The approach to determining drill hole order. Each hole introduced a potentially greater level of disturbance; holes are listed in order of drilling.

Hole	Depth (m)[1]	Distance (m)[2]	Booster[3]	Test rationale
1	–	52 45	–	Pad constructed but not drilled
2	–	31 64	–	Pad constructed but not drilled
5	60	40 122	–	First test of nearby drilling
4	90	38 105	used	Tested for effect of the booster
3	60	25 84	–	Tested for effect of drilling in close proximity to the old workings
7	150	68 129	used ^{M}	Tested for the effect of morning drilling
8	130	55 108	used ^{M}	Tested for cumulative and delayed effects of drilling
6	100	59 143	used ^{M}	Tested for cumulative and delayed effects of drilling

[1] Hole depth;

[2] Distance of the rig from the mine entrance and roost, respectively, see Figure 1;

[3] M: indicates drilling commenced in the morning.

from the booster on the surface; and from the RC drill below the surface. The magnitude of a disturbance from noise or vibration was not measured directly (due to limitations on equipment); rather it was determined indirectly through the response of the bats to the activities. Four types of response were considered possible in both *M. gigas* and *R. aurantia*:

Type 1: Bats emerge from the mine during daylight hours. This response would most likely lead to bat mortality. Subsequently, observations of bats emerging at dusk on the same day would establish whether part of the colony still remained in the mine.

Type 2: Bats become active during daylight hours and fly about within the mine adit, but do not emerge into the open. This type of activity normally occurs before nightly emergence. Pre-drilling observations would establish the time bats normally begin to become active before nightly emergence. This type of response could lead to a *Type 1* response.

Type 3: Bats vacate a roost after dusk emergence and do not return at dawn. This can be assessed from a comparison of counts or presence data before, during and after drilling, and through observations made during drilling.

Type 4: Bats do not enter their normal circadian rhythm after their return at dawn, but also do not fly about within the mine. This type of response was not able to be assessed in the present study.

Five measurements were taken in order to be able to detect both immediate and delayed responses to a disturbance. The measurements were different for *M. gigas* and *R. aurantia*.

Measurement 1: *Presence / absence of R. aurantia at alternative roosts*. The presence of *R. aurantia* at other mines in the local area was assessed prior to drilling. This was done to ensure that mines were present in the local area that were suitable for occupation by *R. aurantia* if drilling caused bats to vacate the Klondyke Queen. Simple presence or absence of *R. aurantia* was assessed after dusk at several disused mine entrances around Marble Bar using electronic bat call detectors: either by placing an Anabat II electronic bat call detector connected to an Anabat delay switch (Titley Electronics) at the entrance to record the period of emergence (two hours following dusk; tapes were later analysed to check for calls of *R. aurantia*), or by visiting the mine with a U30 bat call detector (Ultra Sound Advice; in heterodyne mode, tuned to the frequency of *R. aurantia* at c. 120 kHz; Armstrong 2001; Armstrong & Coles 2007) after counts of *M. gigas* at the Klondyke Queen had concluded (see *Measurement 5*). While electronic bat detectors could not be used to confirm whether *R. aurantia* roosted in a mine, they were the best indicator available of presence given time constraints, and safety issues regarding entry and approach of old mines. Surveys were further complicated by the fact that many of the mines have several entrances.

Measurement 2: *Presence / absence of R. aurantia at the Klondyke Queen mine*. The presence of *R. aurantia* in the Klondyke Queen was assessed using a U30 bat detector in heterodyne mode before pad construction, the period between pad construction and drilling, and at all times during drilling near the mine. All previous surveys at the Klondyke Queen since 1995 (Armstrong 2001) have recorded *R. aurantia* as being present. Therefore, if absence was recorded after pad construction or the commencement of drilling, a disturbance would be assumed. The total number of *R. aurantia* in the mine is difficult to estimate because of the large size of roof exits in the mine, post emergence swarming behaviour, and their propensity to relocate to alternative roosts following their capture. Determining the presence or absence of *R. aurantia* at the mine entrance was undertaken instead. This measurement monitored for a *Type 3* response.

Measurement 3: *Observation of daytime emergence*. The possibility of a *Type 1* response was determined through direct observation during drilling. While the rig was present in the Klondyke Queen project area, an observer was stationed at the adit entrance (Point A; Figure 2) to observe whether bats vacated the mine. These species normally never vacate their roosts during daylight hours due to the risk of exposure, predation and lack of available food.

Measurement 4: Daytime acoustic detection. It was assumed that bats normally exhibit low levels of activity during the day, and only fly about within the structure and approach the entrance not long before their emergence. If calls were detected earlier during drilling compared to the period beforehand, it might indicate a *Type 2* response, or an imminent *Type 1* response. Two pieces of recording equipment were lowered through a hole in the roof to the point where bats emerged from the fully enclosed portion of the adit (Point B; Figure 2):

1. A U30 bat detector, tuned in heterodyne mode to detect the calls *of R. aurantia*, which was monitored with headphones whenever the rig was present in the project area until nightly emergence time of *R. aurantia* (*c.* 17:45). The microphone was connected to the detector via a 10 m extension cord, so the observer could be positioned above Point B (Figure 2). The number of calls of *R. aurantia* was to be tallied over the recording period, but the main function of the U30 was to provide real-time indication of whether this species was approaching the entrance and likely to emerge into daylight, allowing rapid feedback to the drilling crew;

2. An Anabat II detector connected to an Anabat Delay Switch and tape recorder, which was also positioned at Point B prior to and during drilling activity at the Klondyke Queen mine (Figure 2). This detector recorded calls of all four species of bat in the mine using a frequency division system (division ratio 8), and analysis was conducted prior to drilling of the subsequent hole. The number of calls of *R. aurantia* was to be tallied during drilling to quantify the level of bat activity. Calls of *M. gigas* were not quantified because both their echolocation and social calls can be difficult to distinguish from other signals based on their structure.

Measurement 5: Dusk emergence counts of M. gigas. Counts of *M. gigas* were used to determine the effect of the drilling program on that species, but they also represented a proxy for the level of disturbance on *R. aurantia*. The number of *M. gigas* emerging from the mine between 17:45 and 20:00 (the period when most of the bats emerged) was recorded both before and during the drilling program. A small head torch was used to illuminate the bats from behind as they flew out of the mine.

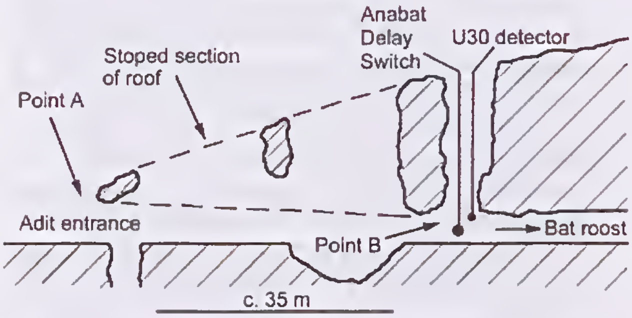

Figure 2. Cut-away section of the Klondyke Queen mine showing the adit entrance, the large gaps in the roof of the mine ('stopes') and where equipment was placed.

Table 3

Emergence times of *R. aurantia* at the Klondyke Queen mine.

Phase	Date	Time	Detection[1]
Pre-bulldozer	12/6/01	17:21	Delay switch
Pre-bulldozer	13/6/01	Prior to 18:30[2]	U30 Point A
Pre-drilling	29/6/01	18:06	U30 Point A
Hole 5	30/6/01	18:04	U30 Point A
Hole 4	1/7/01	17:28	U30 Point B
Hole 3	2/7/01	18:22	U30 Point A
Hole 7	3/7/01	17:28	U30 Point B
Hole 8	4/7/01	17:47	Delay switch
Hole 6	5/7/01	17:53	U30 Point A

[1] The equipment used, and point where observation was made. See Figure 2 for the location of Points A and B;

[2] Equipment problems.

Results

Presence / absence of *R. aurantia* at alternative roosts (*Measurement 1*)

R. aurantia was detected at two other mines near Marble Bar prior to drill pad construction and drilling: Bow Bells (22:30 on 12/6/01) and MacKinnons (within 1 km of the Comet mine; 21:30 on 13/6/01; Figure 1). This species has been observed at the Bow Bells mine on several occasions previously (Armstrong 2001), however, the observation at MacKinnons mine represented a new record. The records suggested that suitable mines were present within the nightly flight range (Bow Bells: 3.9 km; MacKinnons: 20 km) of individuals roosting in the Klondyke Queen mine for relocation if the drilling program caused a disturbance. *R. aurantia* was not detected at the Comet, Golden Gauntlet, Klondyke King, Kopcke's Reward or the Trump mines prior to drill pad construction or drilling.

Presence / absence of *R. aurantia* at the Klondyke Queen mine (*Measurement 2*)

R. aurantia was observed at the Klondyke Queen mine at every nightly emergence prior to and during drilling. Individuals exhibited swarming behaviour between dusk and 20:00, emerging at approximately the same time every evening (*c.* 18:00; Table 3) suggesting that drilling did not cause them to 'hurriedly' vacate the mine after dusk in response to a disturbance (*Type 3* response). Unfortunately, determining whether drilling caused some individuals to relocate during the program was not possible due to the difficulty of obtaining accurate counts of *R. aurantia*. Thus, drilling and pad construction did not cause *R. aurantia* to abandon the mine completely.

Observation of daytime emergence (*Measurement 3*)

An observer stationed at Point A when the drill rig was present did not observe any bat vacating the mine prior to dusk (*Type 1* response). Several *T. georgianus* were observed on walls within the first 40 m of the entrance where the roof is open, however this species normally occupies the semi-dark areas of mines (Jolly 1990), they were observed in this area before drilling, and they did not vacate the mine in daylight.

Table 4

Counts of *M. gigas* between 17:45 and 20:00 at the Klondyke Queen mine. Data on moon position and phase (both in degrees at 18:00) is included for comparison with the progress of the drilling project.

Phase[1]	Date	Number *M. gigas*[2]	Moon Altitude[3]	Moon Azimuth[3]	Moon fraction illuminated
Pre-bulldozer	12/6/01	254	< 0	—	—
Pre-bulldozer	13/6/01	338	< 0	—	—
Pre-drill	29/6/01	108	66.4	47.0	0.64
Hole 5 (60 m)	30/6/01	127	60.0	71.6	0.74
Hole 4 (90 m) B	1/7/01	107	50.9	86.5	0.83
Hole 3 (60 m)	2/7/01	121	40.6	95.9	0.90
Hole 7 (150 m) BM	3/7/01	128	29.9	102.4	0.95
Hole 8 (130 m) BM	4/7/01	106	19.2	107.5	0.99
Hole 6 (100 m) BM	5/7/01	366	8.5	111.6	1.0 (full)

[1] B: indicates booster used; M: indicates drilling commenced in the morning;

[2] Total number counted between 17:45 – 20:00 hours;

[3] Moon position in degrees.

Daytime acoustic monitoring *(Measurement 4)*

The Anabat II – delay switch unit was used to obtain data on the normal emergence time of *R. aurantia*. Before pad construction, *R. aurantia* were heard at Point B (Figure 2) at 17:21. No *R. aurantia* were ever observed or heard before this time during drilling (Table 3) indicating that drilling did not cause a *Type 2* response. No calls of *R. aurantia* were heard on the U30 bat detector during drilling activity, however, *R. aurantia* was heard at 17:28 on two occasions (Table 3). On both occasions, the drill rig had left the project area or had shut down earlier, so their activity was interpreted to be unrelated to drilling activity. Social 'chirp' calls of *M. gigas* were heard frequently between 07:00 and 17:45 at Point B on both the U30 and Anabat detectors, and both before and during the drilling programme, but were not quantified.

Dusk emergence counts of *M. gigas* *(Measurement 5)*

The number of *M. gigas* varied during the monitoring period but this did not seem to be correlated with drilling activity (Table 4; Figure 3). There was a drop of c. 200 individuals after pad construction (consistent with a *Type 3* response). However, by the end of the drilling programme the number of *M. gigas* was greater than prior to pad construction. Colony size fluctuation did not appear to be correlated with obvious natural factors such as moonlight (Table 4).

Discussion

Short term effects of RC drilling

None of the five measurements made during drilling suggested that bats of conservation significance were affected by an RC drilling programme in relatively close proximity to their roost, with both species present throughout the study. The study was designed to detect a range of possible response types, despite a limitation that actual levels of the disturbance from noise and sub-surface vibration could not be measured directly either at the roost or the mine entrance. Rather, the level of potential disturbance was inferred to be dependent on factors such as proximity of the drill rig from the mine entrance, drilling commencement time, and hole depth. The drill rig produced relatively low and steady levels of noise, with surface noise levels increasing during drilling and rod extraction, and decreasing during down-hole inspections of geology. No bat of any species was seen to exit the mine during the day, and audio monitoring did not detect increased levels of diurnal activity. Furthermore, emergence times appeared unchanged and colony size did not show a pattern of gradual or sudden reduction with sustained low numbers. Therefore, there was no evidence to suggest that this particular drilling programme adversely affected mine occupancy or colony size of either *R. aurantia* or *M. gigas* in the Klondyke Queen mine.

A relatively small colony (thought to number approximately ten) of *R. aurantia* appeared to occupy the mine during the program. Obtaining accurate counts of this species without an invasive approach involving trapping or roost entry is not possible, but it was obvious that, for example, a colony an order of magnitude larger in size was not present. Given the small estimated low number of *R. aurantia* present, the Klondyke Queen might initially be regarded as being of little importance to their conservation. However, only around six other sites or localities have been confirmed as supporting significant numbers of this species, which comprises the majority of the known abundance (*e.g.* Armstrong 2001; DEWHA 2010a). There have been numerous new records of bats throughout the region, but no survey has confirmed roosting or established the presence of colonies of significant size (McKenzie & Bullen 2009; K.N. Armstrong unpubl. data). Given the physiological constraints of the species, the security of the regional population is essentially correlated with the number of known roosts rather than the number of records of bats in flight. It is also possible that the Klondyke Queen is used by breeding individuals, though very little is known about their breeding activity in the Pilbara (Armstrong 2001).

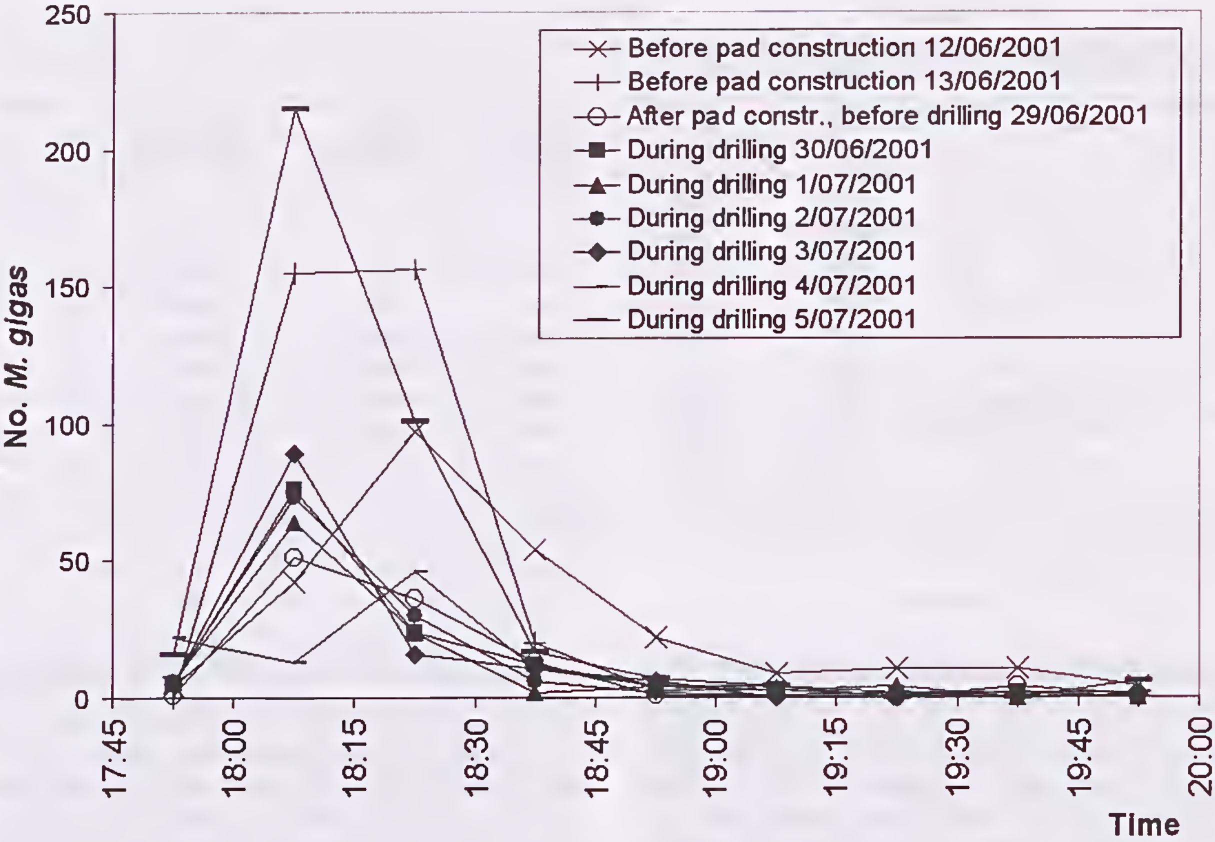

Figure 3. Counts of *M. gigas* at emergence at the Klondyke Queen mine, totalled in 15 minute intervals.

Confounding factors

The counts of *M. gigas* were significantly lower at the Klondyke Queen mine following pad construction, suggesting that the use of a bulldozer might represent a greater disturbance than a drill rig, especially within 50 m of an adit entrance (Hole 3). The bulldozer was operated in close proximity to the old workings to construct a pad for Hole 3. Pad construction for Holes 1 and 2 (not undertaken) closer to the mine entrance might have had greater potential for detrimental effect. However, during the drilling program numbers of *M. gigas* returned to and even exceeded values recorded prior to pad construction. Interpreting the variation in the number of *M. gigas* during the monitoring program is confounded by four possible factors: the activities at the Klondyke Queen mine, including both pad construction by the bulldozer and the operation of drill rig equipment; the (undocumented) concurrent activities at other local mines, in particular the nearest large colony at the Comet mine; moonlight caused bats to stay in the mine during at least part of the night; and any natural patterns of movement that might usually occur. Previous counts of *M. gigas* at the Klondyke Queen recorded 98+ (Hall *et al.* 1997) and 30+ (20/6/96; Armstrong unpublished data in Armstrong & Anstee 2000), suggesting that the number of *M. gigas* has fluctuated previously at the Klondyke Queen. If the short term use of a bulldozer used in pad construction caused some degree of disturbance to *M. gigas*, this was clearly temporary, but it should be highlighted that not every situation might result in a 'recovery' of numbers in the short term. The entrance of the mine did not face the pad for Hole 3, and the result might have been different with a different level of noise. Other disturbances such as roost entry and capture might represent greater levels of disturbance.

Previous observations at the Klondyke Queen mine (K.N. Armstrong unpubl. obs.) suggested that the emergence of *M. gigas* could be delayed by several hours during periods of bright moonlight. This was also noted previously at Mt. Etna, Queensland where *M. gigas* remained in their roost for four nights during the full moon (J. Toop pers. comm. to P. Helman and S. Churchill). No similar observations were made during the present study, despite the presence of bright moonlight. The adit actually faced the direction of moon rise so that the moonlight was shining directly into the mine, and the majority of *M. gigas* exited the mine within a 45 minute period after dusk (Figure 3). Furthermore, a similar number emerged from the mine during periods of both no and full moon. Thus, neither drilling nor phase of the moon had any detectable effect on emergence time, and therefore delayed emergence during periods of bright moonlight might not always occur.

It is suggested that activities occurring at other mines might have had an influence on the number of *M. gigas* at the Klondyke Queen mine. Natural patterns of movement must also be considered, though little is known about their breeding activities and short term or seasonal movements in the Pilbara. It was not possible in this study to determine which other local mine *M. gigas* were moving to, though the Comet mine is the most likely candidate because it is the nearest large permanent colony of this species (Hall *et al.* 1997). The increased numbers at the end of the drilling programme might have resulted from disturbance at a nearby roost such as the Comet. Mine tours were conducted for tourists several times per day at the Comet in 2001, especially in winter. If tours are not conducted appropriately, disturbance to *M. gigas* cause roosting bats to take flight within the mine (K.N. Armstrong pers. obs.), so daytime disturbance in the Comet has the potential to result in higher rates of movement amongst local mines. Concurrent disturbances at nearby major roosts might multiply the effects of individual events.

Possibility of long term cumulative effects

Cumulative effects are a typical cause of an extinction vortex (Gilpin & Soulé 1986). The present study was short-term in duration, however the disturbances associated with drilling, though minor and not detectable by the monitoring approach undertaken, might have the potential to contribute in combination with other factors to longer term effects. Such considerations are relevant in the context of conservation assessments (*e.g.* IUCN Red List criteria A and C; IUCN 2001). Gradual declines in occupancy and colony size might result from a greater frequency of roost switching from a range of factors at multiple local roosts, forced relocation to suboptimal roosts, and interruption of breeding activity. There is some evidence to suggest that *R. aurantia* has moved in the eastern Pilbara more than once in response to a range of disturbances (Armstrong 2001). It is obvious that both past and present (*i.e.* this study) levels of disturbance have not affected usage of the Klondyke Queen mine by *M. gigas*, but numbers of *R. aurantia* do appear much lower than the estimated 350 recorded in 1981 by Churchill *et al.* (1988) (from a capture of 72 individuals). Relatively few individuals have been observed since, though accurate numbers could not be obtained (Armstrong 2001). This is correlated with a range of past activities, including drilling by CRA Exploration Pty Ltd in 1994–1995; the capture of individuals as part of scientific surveys on several occasions; and blasting associated with construction of the Mullan adit (below the old Klondyke Queen workings) sometime in the 1980's. There is also evidence that disturbances in other workings in the district has caused *R. aurantia* to relocate. After a large colony of *R. aurantia* was observed in the Comet mine in 1991 (J.N. Dunlop pers. comm. to N.L. McKenzie; Armstrong 2001), an exodus may have occurred for unknown reasons, and again in 1997 when a pump broke and the lower levels of the mine filled with water. A large proportion of the eastern Pilbara subpopulation now appears to have moved to the Bamboo Creek mine. They were not known from this structure prior to 1996 when it was still operational, however there is no survey data prior to this time. Unfortunately there is only a small amount of data on colony size and mine occupancy in the district (Hall *et al.* 1997; Armstrong 2001), and no information on whether past disturbances have decreased survivorship each time the colony was forced to move.

Conclusions

The present study observed no significant effect of the drilling programme. Furthermore, the drilling programme introduced a level of disturbance that was smaller than that from other activities that have occurred in the past in the Klondyke Queen and nearby eastern Pilbara mines. The present study is detailed so that similar programmes can be designed if drilling or similar activities are considered to have a potentially detrimental effect on colonies of these species. Since this study was undertaken, there is now a greater range of electronic equipment that can be used to monitor bat presence and colony size remotely, including those based on acoustic, video and thermal image recordings. The potential to implement comprehensive and well designed studies using non-invasive approaches (or those that limit disturbance) is much greater, and should be within the capacity of environmental consultants.

If any future drilling (full day, with a booster present) at the Klondyke Queen were to take place further than 25 m from the roost entrance, equivalent to 85 m from the roosting location within the mine, and old workings are not intersected, no significant effect would be anticipated. However, the use of a bulldozer in close proximity of a mine entrance and roost location would need to be considered carefully. These figures might be relevant at other mines, but local conditions would need to be considered.

It is clear that the combined effect of various disturbances will not be known unless there is an organised way of tracking mining related activities and bat occupancy. The creation of a regional plan that can provide guidelines for making assessments of potential disturbance and provide a strategy and recording system for monitoring of contemporaneous impacts would be a first step. Also, when similar issues arise at the Klondyke Queen mine or others in the eastern Pilbara, it might be more intelligent to direct limited resources to actions other than simple monitoring, given the information resulting from the experience of the present study. In the case of the Klondyke Queen mine, it has been known since 1995 when CRA Exploration undertook evaluation drilling, that there were significant reserves of economically viable ore deposit beneath the historical workings. If there was sufficient lead time before the commencement of mining, a more satisfactory result could be achieved for both the miner and the bats, if the proponent, consultant or land manager were able to use the available information to develop a more long term solution. A carefully designed exclusion programme coupled with the preservation, augmentation and / or repair of nearby alternative sites might have a better long term outcome. This is an excellent example of a more broad-scale view that could be gained from a regional plan for management of *M. gigas* and *R. aurantia*.

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Implications of terminal velocity and wing loading on *Hakea* (Proteaceae) seed dispersal

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Abstract

Hakea (Proteaceae) has winged seeds that rely on wind movement for dispersal, with seed dispersal being most effective in a post-fire environment. Seeds from 14 southwestern Australian species (8 fire-killed, 6 resprouters) had their seed mass, wing loading, and terminal falling velocity recorded to determine if a) overall relationships existed between seed properties and seed aerodynamics, and b) if seed dispersal was inherently different between the two post-fire persistence strategies. An increase in seed wing loading, calculated as the mass divided by area of the winged seed, resulted in an increase in seed terminal velocity. Using terminal velocity data to estimated initial seed dispersal distance, *Hakea* seeds have the potential to be dispersed up to 5 m from a parent plant, assuming seeds dispersed from less than 1.5 m above ground, and prevailing wind speeds of no more than 5 m s^{-1}. However, under field conditions *Hakea* seeds are dispersed up to 20 m from the nearest adult. Secondary dispersal, the movement of seeds across the ground, is therefore more important in explaining *Hakea* post-fire dispersal patterns, and is a function of seed mass, prevailing surface wind conditions and the occurrence of litter microsites. Despite detecting no significant difference in wing loading or terminal velocity between fire-killed and resprouting in this study, field observations suggest that seeds of resprouter *Hakea* species tend to disperse their seeds further away from the parent plants.

Keywords: winged seeds, wind dispersal, falling speed, primary and secondary dispersal, seed movement

Introduction

Within the family Proteaceae, wind dispersed seeds mostly occur within genera that possess woody fruit (*i.e. Banksia, Hakea, Xylomelum* from Australia) or woody cones (*e.g. Petrophile* (Australia); *Protea, Leucadendron* (South Africa)). The former producing seeds that possess papery broad wings. The wing provides airborne seeds with greater lift by decreasing the seeds' terminal velocity, defined as the speed at which a seed ceases to accelerate because the downward force of gravity equals the upward force of air drag. For winged objects the terminal velocity is proportional to the square root of the object wing loading, defined as its total mass divided by area (Norberg 1973). A decrease in terminal velocity has implications for longer dispersal distances (Green 1980), potentially kilometres away from the parent plant, when conditions are favourable (He *et al.* 2004, 2009; Schurr *et al.* 2005). However long distance dispersal only accounts for approximately 1–3% of wind-dispersed seeds (Nathan *et al.* 2002; Soons & Ozinga 2005).

Within the genus *Hakea* seed size varies from 2 – 500 mg (mean of 32 mg) (Groom & Lamont 1996) with the wing accounting for at least a quarter of the winged seed area. Seed release can occur within hours of the fruit opening, often in response to the passage of fire or other types of stem damage (McCaw & Smith 1992). Once dispersed the seeds tend to accumulate in litter microsites or become trapped within the existing vegetation (Lamont *et al.* 1993). Most *Hakea* species release their seeds into a post-fire environment (Groom *et al.* 2001) which provides a suitable barren landscape for effective dispersal of wind dispersed seed (Manders 1986; Bond 1988; Yeaton & Bond 1991; Lamont *et al.* 1993; Hammill *et al.* 1998; Schurr *et al.* 2005).

This paper investigates the relationships between seed mass, wing loading and terminal velocity of 14 southwestern Australian *Hakea* species and the implications of these traits on seed dispersal following fire. Fire-killed and resprouter species were selected to investigate whether there is any functional significance in dispersal traits that relates to differences in post-fire persistence strategies.

Methods

Seed collection

Approximately 20 fruits per species were harvested from natural populations of 14 *Hakea* species from locations near Eneabba (*H. circumalata, H. incrassata, H. smilacifolia*), Kalamunda (*H. amplexicaulis, H. erinacea, H. cyclocarpa, H. lissocarpha, H. undulata*), the Stirling Ranges (*H. lehmanniana, H. pandanicarpa* subsp. *crassifolia, H.*

prostrata, H. trifurcata) and Watheroo (*H. psilorrhyncha, H. cygna* subsp. *cygna*), Western Australia. Location details are provided in Groom & Lamont (1995). Species were co-occurring at each location, and were selected to ensure that a range of fire-killed and resprouting species were represented. The woody fruit were allowed to air dry for two weeks in a glasshouse to enable the fruit to naturally dehisce. Once open, seeds were carefully extracted using forceps to ensure the removal of entire diaspores (seed plus wing). Enough fruit opened to obtain 20 intact seeds per species.

Seed properties

For each species the fresh mass and projected area of 20 winged seeds, and the fresh mass of the seed only were measured. Mass was measured using a 4 decimal place balance. Projected areas (*i.e.* silhouette of each winged seed) were measured using a digital imaging system (DIAS-II, Delta T Devices, UK) that utilised a digital video connected to a computer to capture, and calculate the area of seed images. The loading on the winged seed (known as 'wing loading') is calculated as the mass divided by area of the winged seed (Matlack 1987).

Falling speed and terminal velocity

Winged seeds for each species were dropped one at a time from a windless indoor height of 4.5 m onto white sheeting. Two observers recorded the time taken for each winged seed to reach the floor using digital stop watches, and the mean fall time calculated. Seeds were retimed if the two values differed by more than 0.1 s. A tenth of a second represents an error in terminal velocity of 5% or less (Marchetto *et al.* 2010). Falling speed was measured as the drop height (h) divided by the fall time (t), and is also known as initial terminal velocity (Greene & Quesada 2005). True terminal velocity (v_f) needs to take into account that a falling seed has an initial period of acceleration, and was calculated according to Greene & Quesada (2005) as:

$$v_f = \frac{gt - \sqrt{g^2t^2 - 2.773gh}}{1.386} \text{ .... eqn 1}$$

where g is acceleration due to gravity (9.81 m s^{-1}), t is elapsed time (s), and h is release height (m)

Estimation of dispersal distance

Predicted dispersal distances of seeds (d) for each species was calculated as follows (after Cremer 1977):

$$d = \frac{w.z}{v_f} \text{ .... eqn 2}$$

where w = wind speed (m s^{-1}) and z = height within the canopy (m). A release height of 1 m and a wind speed of 5 m s^{-1} were used to allow for comparisons with other data sets (*e.g.* Hammil *et al.* 1998). The latter is comparable with wind speeds recorded at the closest meteorological stations to the collection sites (Bureau of Meteorology, Western Australia). Distances were also calculated using a release height of 1.5 m, reflecting the mean canopy height of the study species.

Results

Correcting for gravitation acceleration (as per Eqn 1) resulted in 3–26% increase in terminal velocity from the original falling speeds (Table 1). Winged-seed area was linearly correlated with mass for winged seeds (r = 0.89, P < 0.0001; Figure 1a) with seed terminal velocity increased linearly with square root of the resultant wing loading (r = 0.65, P = 0.012; Figure 1c). There was no relationship between winged-seed mass and terminal velocity (r = 0.12, P = 0.689; Figure 1b) or winged-seed mass and wing loading (r = -0.13, P = 0.965). The wing increased the weight of the diaspore by 0.1 g (*H. smilacifolia*) to 15.7 g (*H. pandanicarpa*), representing 0.7% (*H. smilacifolia*) to 14% (*H. trifurcata*) of the entire diaspore.

Seeds released 1 m above the ground have the capacity to be dispersed 1.2–3.2 m at a wind speed of 5 m s^{-1}, with an average of 2.4 m (Table 1). Releasing seeds at 1.5 m extends the average distance to 3.6 m. Calculated dispersal distance decreased linearly with the square root of wing loading (r = -0.644, P = 0.013; Figure 1d) and hence with terminal velocity (r = -0.962, P < 0.001). There was no significant difference in seed dispersal properties or calculated dispersal distances between fire-killed and resprouter species (Table 1).

Discussion

Seed wind dispersal comprises of two phases. Primary dispersal involves the movement of seed through the air, with the end result that the winged seed is deposited on the ground. For *Hakea* species, this requires the seeds to be blown out of the open woody fruit. Seeds may then be blown across the ground until they are permanently trapped by leaf litter, organic debris and existing or dead vegetation, or until the membranous wing is damaged. This is known as secondary dispersal, and has more direct implications on post-dispersal seedling recruitment patterns (Bond 1988; Chambers & MacMahon 1994; Hammill *et al.* 1998; Schurr *et al.* 2005) as it is affected by seed traits, prevailing ground wind conditions and barriers to seed movement (Johnson & Fryer 1992; Schurr *et al.* 2005). The duration of wind assisted seed movement is dependent on when a seed becomes entrapped by surface obstacles or covered by soil, or when the first heavy rains occur.

Primary dispersal is directly linked to the winged seed's terminal velocity and is a linear function of seed wing loading, providing the winged seed acts as a drag device (Greene & Quesada 2005). Previously published terminal velocity values for *Hakea* seeds range from approximately 1 m s^{-1} (Richardson *et al.* 1987, Hammill *et al.* 1998) to a maximum of 2.1 m s^{-1} (Lamont *et al.* 1993), the latter being within the range of speeds presented in this paper (Table 1). Differences between datasets (*i.e.* lower values for eastern Australian hakeas (Hammill *et al.* 1998) and higher values western Australian hakeas (Lamont *et al.* 1993; this paper)) may imply that eastern Australian hakeas are more likely to disperse over greater distances. Experimental data collected by Hammill *et al.* (1998) suggests that *Hakea* seeds with an average wing loading of 220 g m^{-2} will have a primary dispersal distance between 5–6 m, almost twice that predicted from this study at

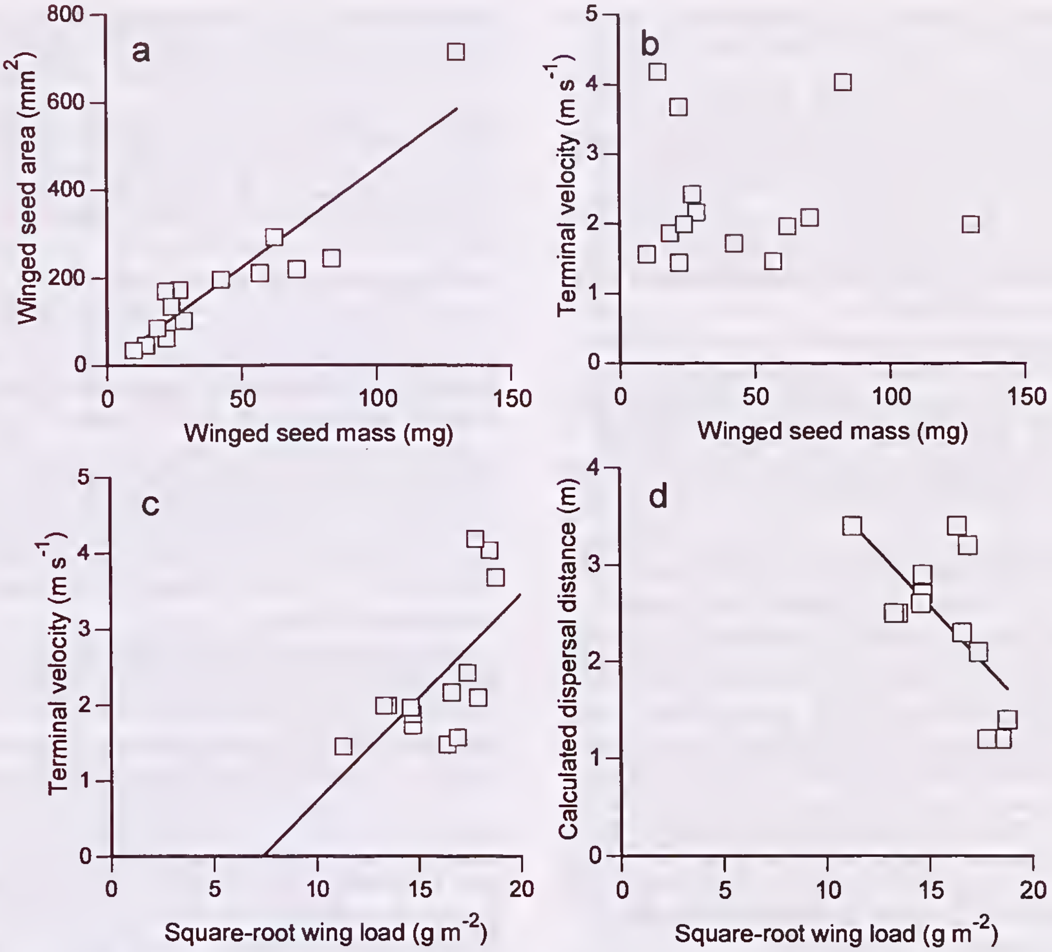

Figure 1. Relationships between winged seed morphology, terminal velocity and calculated primary dispersal distances (from 1 m release height) for 14 *Hakea* species. Trend lines are provided for significantly correlated data. See Table 1 to identify data for individual species, and hence species' contribution to overall trends.

Table 1

Seed traits and seed dispersal properties of 14 *Hakea* species. Dispersal distance calculated for all species assuming a release height of either 1 or 1. 5 m and a wind speed of 5 m s^{-1}. Data mean ± SE. Trait comparison between fire-killed and resprouter conducted using *t*-tests.

Species	Seed mass (no wing) (mg)	Winged Seed Area (cm^2)	Wing loading (g m^{-2})	Fall speed (m s^{-1})	Terminal Velocity (m s^{-1})	Dispersal distance (m) 1 m	1.5 m
Fire-killed							
H. circumalata	26.5 ± 1.8	1.35 ± 0.07	176.5	1.88 ± 0.04	1.99 ± 0.05	2.5	3.8
H. cygna subsp. *cygna*	25.3 ± 1.2	1.03 ± 0.02	276.7	2.01 ± 0.08	2.16 ± 0.11	2.3	3.5
H. erinacea	21.7 ± 0.7	0.46 ± 0.01	315.2	3.28 ± 0.03	4.19 ± 0.07	1.2	1.8
H. pandanicarpa subsp, *crassifolia*	114.7 ± 6.7	7.15 ± 0.53	182.4	1.87 ± 0.05	1.99 ± 0.07	2.5	3.8
H. psilorrhyncha	65.8 ± 7.5	2.20 ± 0.10	320.2	1.95 ± 0.05	2.09 ± 0.07	2.4	3.6
H. smilacifolia	9.8 ± 0.6	0.34 ± 0.01	287.0	1.50 ± 0.04	1.56 ± 0.05	3.2	4.0
H. trifurcata	15.9 ± 1.3	0.85 ± 0.03	217.5	1.77 ± 0.11	1.87 ± 0.13	2.7	4.0
H. undulata	18.9 ± 0.4	0.62 ± 0.01	353.6	3.02 ± 0.08	3.68 ± 0.19	1.4	2.0
Mean	37.3 ± 12.5	1.75 ± 0.80	266.1 ± 23.5	2.16 ± 0.32	2.44 ± 0.34	2.3 ± 0.2	3.3 ± 0.3
Resprouters							
H. amplexicaulis	31.1 ± 0.9	1.97 ± 0.13	215.3	1.67 ± 0.04	1.72 ± 0.04	2.9	4.4
H. cyclocarpa	80.5 ± 2.3	2.44 ± 0.11	341.2	3.21 ± 0.04	4.04 ± 0.08	1.2	1.8
H. incrassata	49.2 ± 3.3	2.92 ± 0.17	212.8	1.85 ± 0.05	1.96 ± 0.07	2.6	3.8
H. lehmanniana	17.5 ± 0.6	1.70 ± 0.08	127.9	1.40 ± 0.07	1.45 ± 0.07	3.4	5.2
H. lissocarpha	32.3 ± 0.6	1.13 ± 0.04	303.8	2.19 ± 0.24	2.42 ± 0.29	2.1	3.1
H. prostrata	34.9 ± 1.0	2.11 ± 0.06	268.6	1.42 ± 0.10	1.47 ± 0.11	3.4	5.1
Mean	40.9 ± 8.9	2.04 ± 0.25	244.9 ± 31.0	1.96 ± 0.28	2.18 ± 0.40	2.6 ± 0.3	3.9 ± 0.5
Fire-Killed vs Resprout (*P* values)	0.314	0.164	0.830	0.998	0.884	0.553	0.553

similar release height and wind speed. These values will underestimate the maximum possible primary dispersal distance under field conditions, as primary dispersal distance increases as a function of air turbulence (Greene & Johnson 1992; Nathan *et al.* 2002) and release height. Terminal velocity thus has limited explanatory value in predicting the final dispersal distance within the genus, as wind exposure, microsite availability and soil texture of the surface on first landing are all important external factors influencing secondary dispersal distances of *Hakea* seeds (Lamont *et al.* 1993).

Winged *Hakea* seed can travel up to 8 m at a constant wind tunnel speed of 6.5 m s^{-1} with the potential to travel more than 12 m in a post-fire environment depending on post-fire wind conditions (Hammill *et al.* 1998). This is similar to the seedling recruitment pattern observed in the field for southwestern Australian hakeas (P. Groom, *pers. observ.*) with seedlings tending to occur within a 5–20 m radius of a parent plant. Seedlings of fire-killed *Hakea* species are more likely to occur closer to an adult, as they recruit more seedlings post-fire than expected by chance alone (Groom *et al.* 2001), resulting in a relatively large number of seedlings recruiting per pre-fire adult. This complements the findings of studies on wind dispersed South African *Protea* species where most seedlings are found in greatest abundance directly underneath the parent plants and lowest in the open, with 95% of seedlings occurring within 15 m of the parents following fire (Manders 1986; Yeaton & Bond 1991). A maximum distance of 26 m from the parents (Manders 1986) implies that the movement of seeds rolling over the soil surface is an important determiner of their final resting distance (Bond 1988).

Fire has a direct impact on secondary seed dispersal, creating the litter (burnt leaves, stems, fruit parts) and burnt vegetation that become the traps for tumbling seeds, with the rate of post-fire litter accumulation and litter microsite formation increasing with time since fire (Lamont *et al.* 1993; Denham *et al.* 2009). In a fire-prone landscape, seeds are often initially dispersed into a barren environment, where litter is relatively scarce (Denham *et al.* 2009). With wind assistance winged seeds can travel up to 4 m in 1 hour (Hammill *et al.* 1998). Secondary dispersal thus is an important consideration when investigating post-fire dispersal patterns of wind-dispersed seed (Schurr *et al.* 2005), with winged seed density in these litter microsites being 18–39 times greater than adjacent bare patches (Lamont *et al.* 1993). Hammill *et al.* (1998) found that *Hakea* seeds moved smaller distances along the ground than *Banksia* seeds (due to their larger seed wing loading) increasing their resistance to movement by wind, and differences in seed morphology. The proportion of the winged seed that is seed (embryo plus testa) is important in understanding the secondary surface dispersal of hakeas.

Wind dispersed seeds of serotinous South African Proteaceae do not disperse as far as their Australian counterparts (Manders 1986; Bond 1988; Hammill *et al.* 1998; Yeaton & Bond 1991), despite plumed and parachute seeds (mostly South African Proteaceae) being more mobile than winged seeds (mostly Australian Proteaceae) (Schurr *et al.* 2005). Recent demographic studies have demonstrated the potential for winged *Banksia* seeds to travel up to 2.6 km from the parent population in the Eneabba sandplains (He *et al.* 2004; 2009). Similar distances have been noted for invasive *Pinus* (pine) species, all of which have winged seeds (van Wilgen & Siegfried 1986). To be transported such long distances, wind updrafts and vortices come into play, as long-distance dispersal is directly related to seed uplifting (Nathan *et al.* 2002). Lamont and colleagues (Lamont *et al.* 1993; He *et al.* 2004; 2009) have suggested that willy-willies (also known as 'dust devils') and prevailing winds play an important role in long-distance dispersal in a post-fire landscape, having the ability to transport soil and surface debris to heights of hundreds of metres (Oke *et al.* 2007). It would be expected that *Hakea* species with smaller wing loads, and hence lower terminal velocities (*e.g. H. circumalata*), would remain within the convective vortex for longer time periods. Thus having greater potential for long-distance dispersal. Although this is dependent on the assumption that the papery wing of *Hakea* seeds does not sustain any significant damage during the process. Removing the wing causes a 12–17% increase in terminal velocity (Lamont *et al.* 1993).

There was no difference in any of the dispersal-related seed properties measured in relation to post-fire survival strategies. It might be expected that seeds of fire-killed hakeas would not disperse as far as resprouter species because of the lack of seedling competition with adults. Although the current study showed no significant difference in seed mass between the two strategies, Groom & Lamont (1996) found that resprouting hakeas tend to produce heavier seeds without increasing wing loading. Experimental work suggests that in terms of secondary wind dispersal, large seeds require higher wind speeds to move across a surface (Johnson & Fryer 1992), and hence one might expect seeds of fire-killed hakeas to quickly move across the ground after their initial dispersal from the parent plant. Field observations suggest that obstacles (*i.e.* accumulated litter) and localised air turbulence just above the ground surface play an important role in seed dispersal patterns within *Hakea* populations, as has been modelled for wind dispersed South African Proteaceae (Schurr *et al.* 2005). The ability to protect and retain viable quantity of canopy stored seeds between fires is more important in explaining differences in post-fire recruitment patterns of co-occurring fire-killed and resprouting *Hakea* species (Groom *et al.* 2001).

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Aggregation and stranding of elongate sunfish (*Ranzania laevis*) (Pisces: Molidae) (Pennant, 1776) on the southern coast of Western Australia

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Abstract

Despite a global distribution and moderate abundance in temperate and tropical seas, the biology of *Ranzania laevis* (Family: Molidae) (Pennant, 1776) is very poorly understood. In autumn/winter 2008, numerous schools of *R. laevis* appeared on the south coast of Western Australia and several hundred individuals were washed ashore. Many of these fish were alive or recently deceased when found, creating a rare opportunity to study this species. Data on length, weight, diet and reproductive status obtained from stranded fish during this unusual event is described in the context of previously published information about this species. All fish examined appeared to be adults ranging from 300 to 700 mm total length. The ratio of males to females was 2:1. Gonad development indicated spawning was not imminent. The cause of the strandings was unclear, although advection of fish to the south coast by the tropical Leeuwin Current was strongly implicated.

Key words: slender sunfish, *Ranzania laevis*, fish kill, southern Western Australia

Introduction

Ranzania laevis (Family: Molidae), or elongate sunfish, has a worldwide distribution in temperate and tropical seas as far north as Scandinavia and as far south as New Zealand (Fraser-Brunner 1951; Hutchins 2001). It is an oceanic species found at depths of 0 m to >300 m (Leis 1977; Watson 1996). The biology and ecology of *R. laevis* is poorly understood, presumably because individuals have an inaccessible distribution and are rarely observed near land, and also because they are not fishery targets and have low commercial value. Existing knowledge of *R. laevis* has been derived mainly from individuals found stranded on beaches or caught accidently in deep ocean fishing operations (*e.g.* Fitch 1969; Ebenezer & Joel 1984; Iddison 2002). As a consequence, many of the published descriptions of this species have been based on specimens that were decomposed, damaged and/or of obscure origin.

Ranzania laevis is often reported as being solitary but, unlike other molid species, adults and juveniles also frequently aggregate (Fraser-Brunner 1951; Robinson 1975; Hutchins 2001). Schools comprising >500 individuals have occasionally been observed (Castro & Ramos 2002). The factors influencing aggregating behaviour are unclear. Aggregations are sometimes associated with warm water masses, although this behaviour could also be a response to be prey availability (Castro & Ramos 2002). *Ranzania laevis* consume small fish and planktonic invertebrates and so are probably attracted to areas of high productivity, such as upwelling and convergence zones, where these prey are abundant (Fitch 1969; Robinson 1975; Dulcic *et al.* 2007).

Individuals and small aggregations of *R. laevis* have previously been reported in coastal waters of south-western Australia (Walker 1983; Gomon *et al.* 2008). This region is strongly influenced by the tropical Leeuwin Current, which flows southward along the west coast throughout the year. In autumn/winter (April–August), the current strengthens and extends southwards along the full length of the west coast and then eastwards along the south coast (Smith *et al.* 1991). This process transports warmer water and various pelagic species to the more temperate region of the south coast (Maxwell & Cresswell 1981; Caputi *et al.* 1996; Gopurenko *et al.* 2006). Transport potential varies according to substantial interannual variations in current strength. The Leeuwin Current is typically stronger during a La Niña year and weaker during an El Niño year (Feng *et al.* 2003).

In April–June 2008, numerous schools of *R. laevis* appeared on the south coast of Western Australia and several hundred individuals washed ashore. Many of these fish were alive or recently deceased when found, creating a rare opportunity to obtain biological information about this species. This paper summarises information on length, weight, diet and reproductive status obtained from stranded fish in 2008 and briefly describes the environmental conditions occurring at the time of, and possibly contributing to, this unusual event.

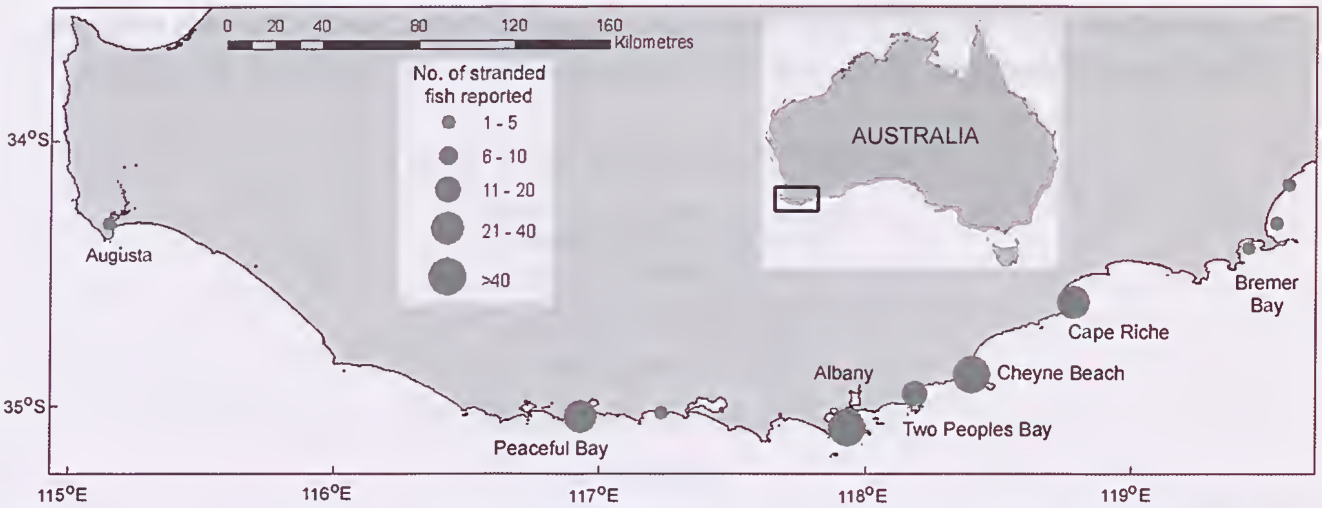


Figure 1. Locations and total numbers of *Ranzania laevis* sighted by researchers and the public between February and June 2008 along the south coast of Western Australia.

Methods

Trends in regional ocean temperature since 1982 were available from satellite-derived monthly mean sea surface temperature (SST) for the 1-degree latitude x 1-degree longitude block containing Albany (*i.e.* 35–36° S, 117–118° E) (Reynolds & Smith 1994). Sea surface temperature images were downloaded from http://www.marine.csiro.au/remotesensing. Intermittent records of beach and inshore water temperature for the Albany region were available from unpublished Department of Fisheries data.

Mean annual sea level at Fremantle, Western Australia, was used as an index of annual Leeuwin Current strength. There is a strong linear relationship between Leeuwin Current volume transport and sea level deviation at Fremantle (Feng *et al.* 2003).

Following initial sightings of stranded *R. laevis* in early May 2008, two relatively fresh fish were collected from Cheyne Beach near Albany (Figure 1) and examined by a pathologist to determine a cause of death. Fish were examined for evidence of pathogens, parasites, poisoning or other injury. Thin sections of heart, gonad, gill, spleen and liver were examined histologically. Lengths and weights of these fish were not recorded.

A media campaign was implemented in late May 2008 to encourage members of the public to report sightings of sunfish. The public was asked to report the number, date, location and total length of each fish observed.

During May and June 2008, specimens were collected for biological analysis by researchers and by the public. Some biological data was not obtainable from decomposed or damaged fish. Morphological variation was clearly evident during initial observations of fish and so measurements of total length, maximum height (*i.e.* distance between tip of dorsal fin and tip of anal fin), height at insertion point of pectoral fin, height at clavus (*i.e.* distance between base of dorsal fin and base of anal fin) and whole body weight were taken from each fish in an attempt to quantify this variation.

Gonads were extracted and weighed. Gender was determined from macroscopic examination and a macroscopic gonad developmental stage was assigned based on the scheme of Laevastu (1965) where stage 1 = immature virgin; 2 = maturing virgin or recovering spent; 3 = developing; 4 = developed; 5 = mature; 6 = spawning; 7 = spent. All gonads were found to be at a similar macroscopic stage. Ovaries from three relatively fresh specimens (total length range 510–550 mm) were placed in 10% formalin solution for several weeks and then examined microscopically (histologically) to corroborate the macroscopic staging. A transverse section was removed from the middle of one lobe of each preserved ovary and processed using standard histological techniques to produce a 7 µm section, which was stained with Harris's haemotoxylin and eosin prior to microscopic examination. Ovarian development stages were assessed microscopically from the identification of the most developed oocytes (Yamamoto *et al.*, 1965; Wallace & Selman, 1981).

Stomachs were retained from 19 relatively fresh fish, collected between 13 and 16 May 2008 from Frenchman Bay and Cheyne Beach near Albany (Figure 1). Stomachs were preserved in 10% formalin. Stomach contents were later extracted, identified and weighed.

Samples of dorsal fin from 20 fish were retained and stored in 90% ethanol for future genetic analyses (not examined in this study).

In an attempt to estimate age, the inner ears from three fish were extracted and examined under a dissecting microscope for otoliths. However, no otoliths were present. Some microscopic calcareous granules were found, which were similar in appearance to the otoconia found in *Mola mola* (Thompson 1888).

Results

Ocean conditions

Regional mean monthly ocean temperature was 19–20 °C from April to June 2008, which was well above the long-term average for each month (Figure 2). Relatively high regional water temperatures coincided with the presence along the south coast of the Leeuwin

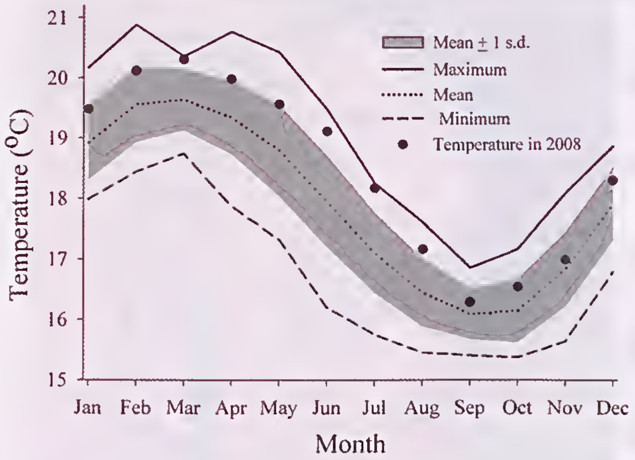


Figure 2. Satellite-derived maximum, minimum and mean (± 1 standard deviation) monthly sea surface temperatures in the Albany region from 1982 to 2007 and mean monthly sea surface temperatures for 2008.

Current. Mean annual current strength in 2008 was the highest since 2000 and one of the strongest years on record (Figures 3 and 4).

At Albany, beach and inshore (*i.e.* up to 10 km from shore) water temperatures were 19–20°C in both April and June 2008 (K. Smith, unpubl. data; N. Chambers, pers. comm.). No records of beach water temperature were available for May. Local residents and commercial fishers reported that beach water temperatures were 'unseasonally warm' at the time of strandings.

Pathology

There was no evidence of any infectious disease or injury causing the death of stranded fish. There appeared to be no major physical damage to any stranded fish, including the gills, although there were some scratches on the skin consistent with abrasion on sand or reef. No external parasites were reported on any stranded fish.

There were no significant histological abnormalities found in sections of heart, gonad, gill and spleen (J. Creeper, pers. comm.). One fish had an enlarged spleen relative to the other fish examined. Also, infections of the wall of the gastro-intestinal tract by encysted metacestodes and an inflammation of the small intestine associated with myxosporidian protozoans were found.

The stomachs of fish examined by pathologists were relatively empty. The appearance of the liver in these fish was suggestive of hepatic lipidosis, where lipids are mobilised in the liver in response to a prolonged period without adequate food (J. Creeper, pers. comm.) (Figure 5A).

Distribution of fish

Between February and June, a total of 322 *R. laevis* were sighted by members of the public and researchers along 520 km of coastline on the south coast of Western Australia, between Augusta and Point Anne (east of Bremer Bay) (Figure 1). The fish occurred as individuals or in small schools (<50 fish) and were observed whilst either stranded on a beach or swimming in shallow waters immediately adjacent to the shore (*i.e.* within 500 m of the shore). The majority of sightings occurred at Albany (40%), Cheyne Beach (37%), Cape Riche (11%), Peaceful Bay (7%) and Two Peoples Bay (4%).

With the exception of a single fish observed on 15 February at Augusta, all fish reported in 2008 were observed between 7 April and 24 June 2008, with the majority (89%) of reported sightings occurring in May. No reports were received of fish swimming in waters further offshore during this period.

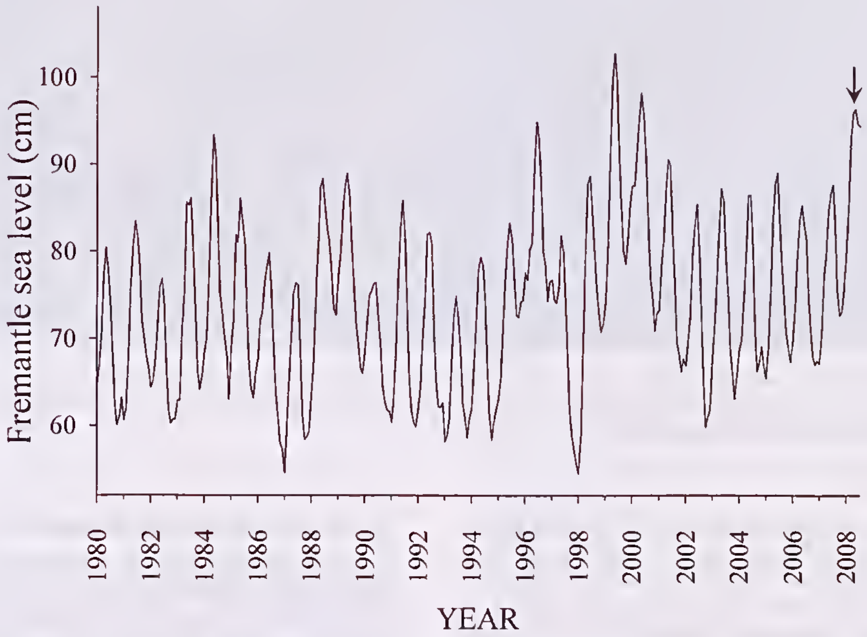
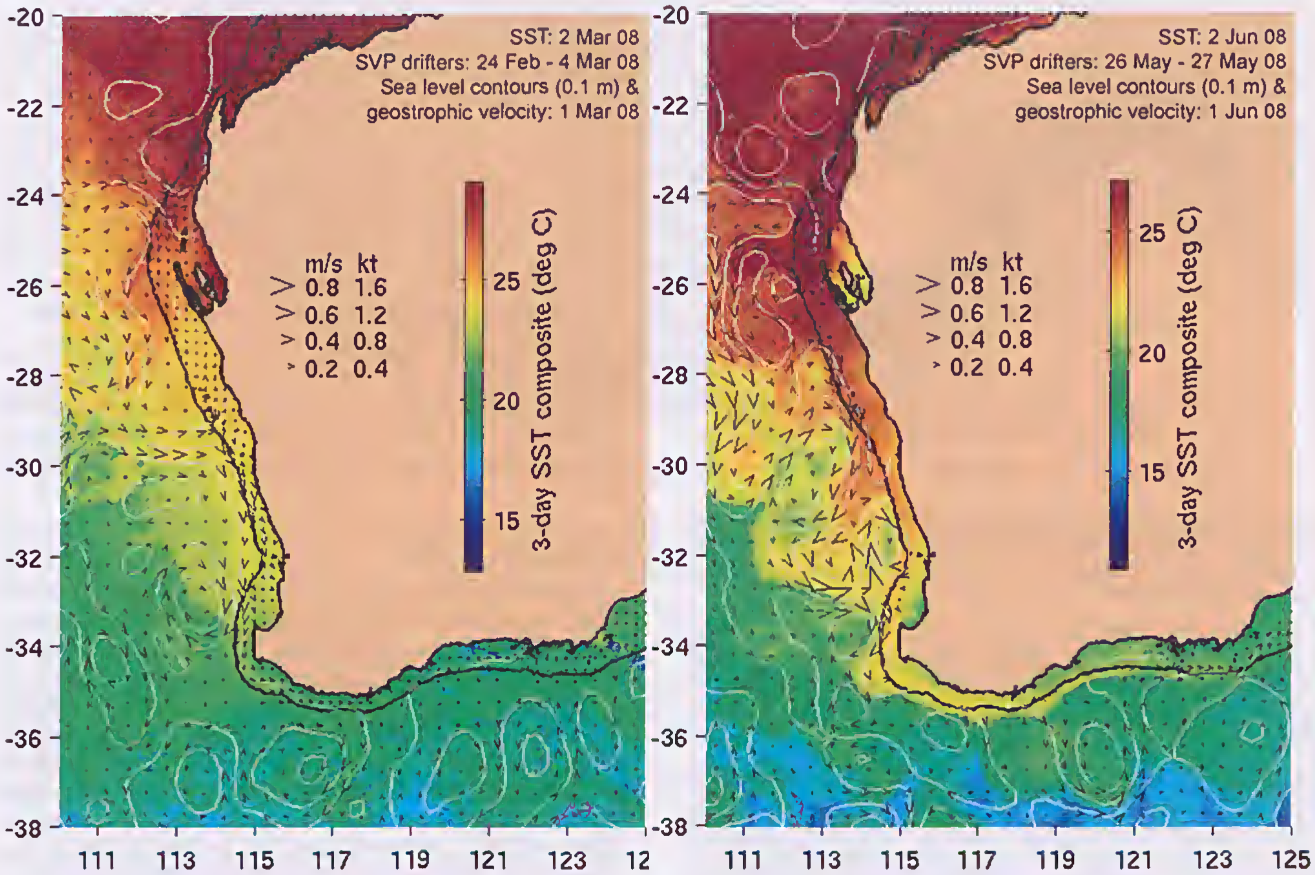


Figure 3. Mean monthly sea level at Fremantle, Western Australia, from 1980 to 2008 (3-month moving average). Fremantle sea level is an index of Leeuwin Current strength. Timing of strandings in 2008 indicated by arrow.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Sea level contours (0.1 m) &
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m/s kt
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3-day SST composite (deg C)
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20
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SST: 2 Jun 08
SVP drifters: 26 May - 27 May 08
Sea level contours (0.1 m) &
geostrophic velocity: 1 Jun 08
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Figure 4. Sea surface temperature images indicating position of Leeuwin Current on A) 2 March and B) 2 June 2008. Images downloaded from http://www.marine.csiro.au/remotesensing.

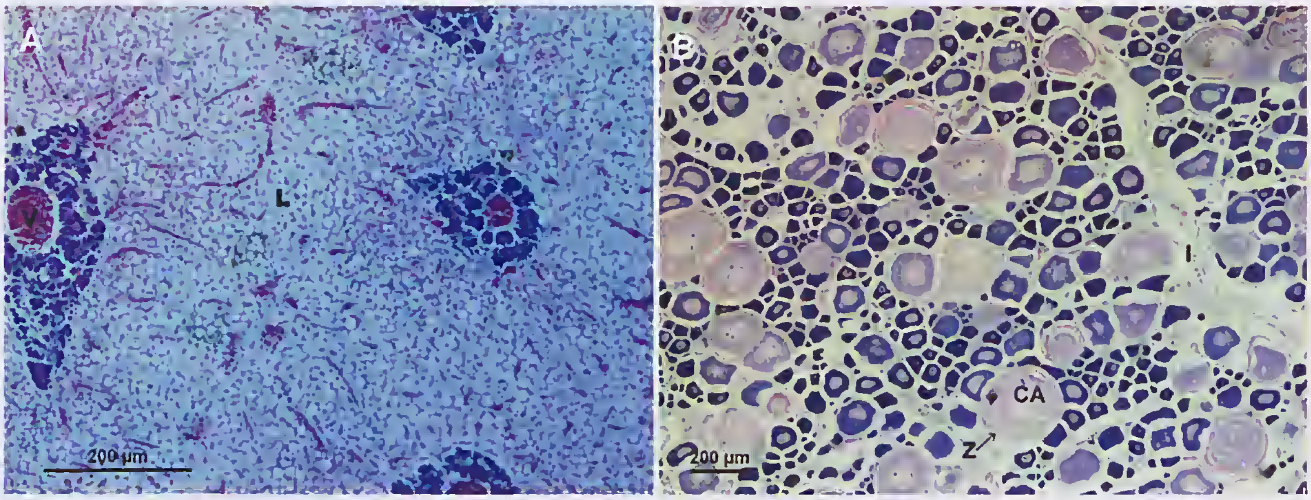

Figure 5. Histological sections from stranded *Ranzania laevis* sampled in May 2008. A) liver displaying abundant lipid vacuoles (L) (clear, spherical); blood vessels (V) also evident. B) ovary showing cortical oleoli (CA) stage oocytes, including well developed zona radiata (Z), and intralamellar stromal strands (I).

In addition to the above sightings, a commercial fisher observed a large school, estimated to comprise 300–350 fish, which swam in shallow waters off Cheyne Beach for 3–4 weeks in early June 2008. None of these fish became stranded while being observed. Overall, this school and the above mentioned stranded fish comprised nearly 700 observed individuals.

Length, weight and sex composition

The total lengths of 95 stranded fish were measured. Of these fish, it was possible to determine the gender of 39 individuals. Gender could not be determined in badly decomposed fish and was not known from specimens reported by the public. The ratio of identified males to females was 2:1. The lengths of stranded fish were

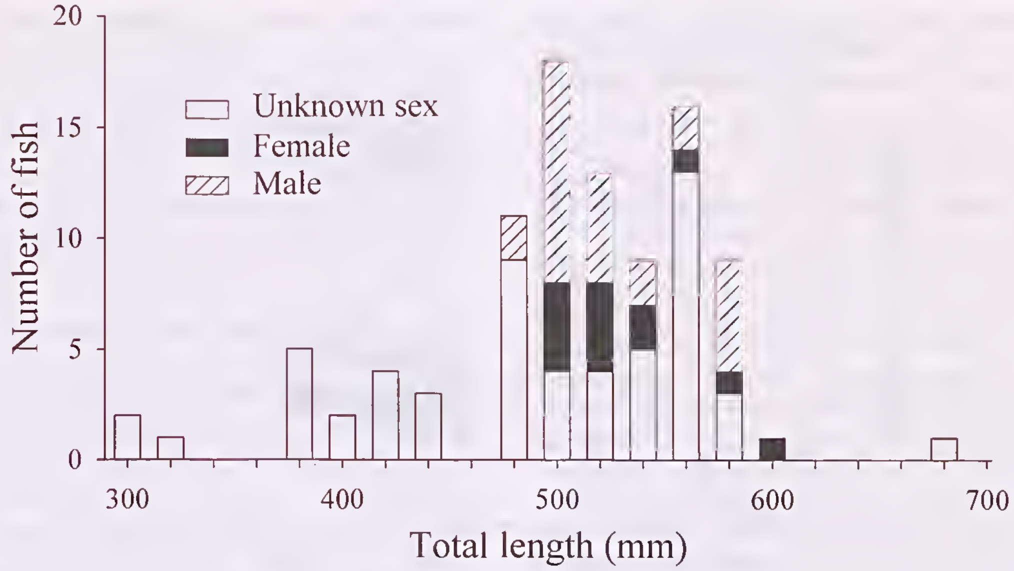

Figure 6. Length frequencies of *Ranzania laevis* sampled between February and June 2008 along the south coast of Western Australia.

reported to range from 300 to 700 mm (Figure 6). Identified males ranged from 490 to 590 mm and females from 506–619 mm.

Relatively few (n=22) undamaged, fresh fish were available to be weighed. The lengths of these fish ranged from 498 to 619 mm. Over this limited range, there was a linear relationship between length and whole body weight:

$W = 0.3412\ L + 374.93$ (where W is weight in g, L is total length in mm).

With the addition of previously published values for similar and smaller sized individuals of *R. laevis*, the combined data suggested the following relationship:

$W = 74.369 \times L^{0.321}$ (where W is weight in g, L is total length in mm) (Figure 7).

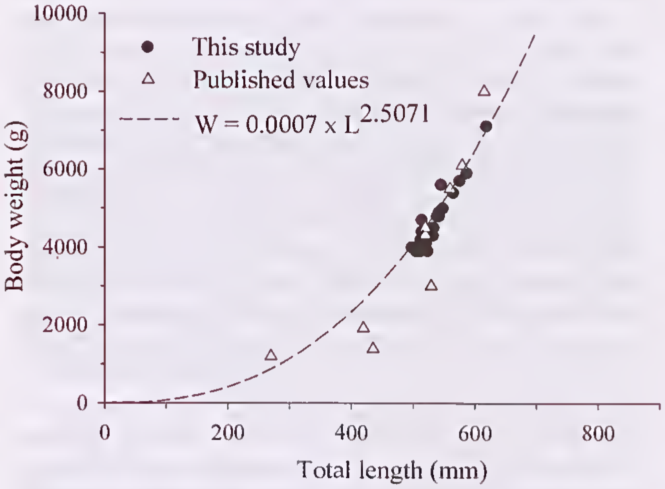

Figure 7. Total length versus total weight of *Ranzania laevis* sampled in 2008 along the south coast of Western Australia and other published values for this species. Data from males and females are pooled. Sources of published values: Specchi & Bussani 1979; Jardas & Knezevic 1983; Walker 1983; Ebenzer & Joel 1984; Iddison 2002.

Over the length range of fish examined in this study, there was clear evidence of an ontogentic change in body dimensions. Small fish were considerably more elongate than larger fish (Figure 8). Body height at the insertion point of the pectoral fin increased from approximately 40% of the total body length at 300 mm to 50% at 650 mm. Maximum height (distance between tips of dorsal and anal fins) increased from approximately 70% of the total body length at 300 mm to 90% at 650 mm.

Reproductive status

Where gender could be determined, all individuals were found to be sexually mature. With the exception a single male fish at stage 2, all other gonads were found to be at macroscopic developmental stage 3 (100% of females and 96% of males). At stage 3, male and female gonads weighed between 0.53 and 1.37% of whole body weight. The weight of the stage 2 testes was 0.02% of whole body weight.

Examination of the microscopic characteristics of ovaries identified the most advanced oocytes in all three specimens as having formed cortical alveoli (CA) and lipid droplets within the cytoplasm (Figure 5B). This oocyte stage indicated that these were Stage 3 'developing' ovaries and that they were dissected from mature females that were in the early stages of commencing vitellogenesis. The identification of well-formed zona radiata for several CA oocytes further confirmed this development stage, although relatively poor preservation obscured any more advanced vitellogenic features within the cytoplasm of these oocytes (Figure 5B). Multiple oocyte development stages indicated asynchronous oocyte development, which suggested that females of this species are serial batch spawners. The maximum oocyte diameter in each ovary ranged from 210 to 274 µm. The presence of broad intralamellar stromal strands in two ovaries indicated that those fish had spawned previously (Figure 5B).

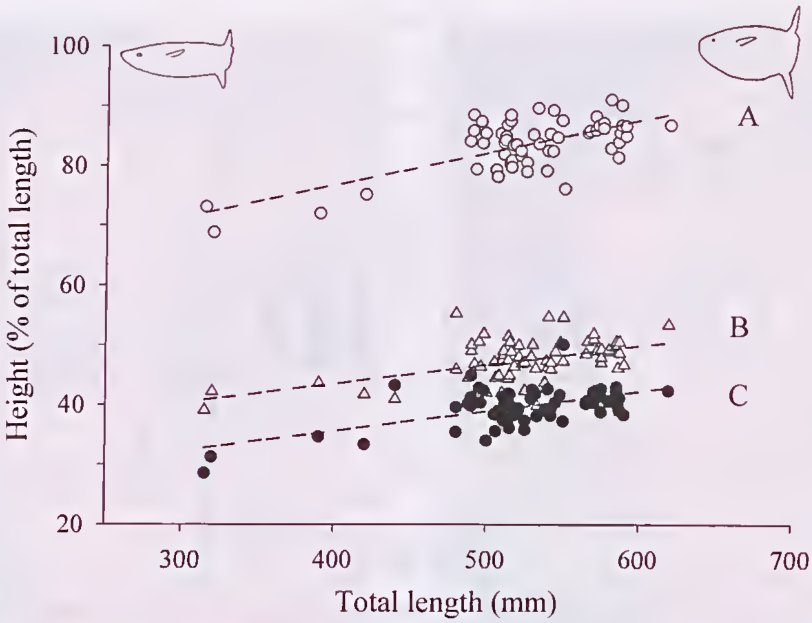

Figure 8. Total length versus A) maximum height, B) height at insertion point of pectoral fin, and C) height at clavus, expressed as a percentage of total length. Measurements by Raven (1939) from a single fish (315 mm) previously collected at Albany are also included. Regressions fitted A) Height = (0.0539 x Length) + 55.105 (r^2 = 0.4708); B) Height = (0.0314 x Length) + 30.925 (r^2 = 0.2445); C) Height = (0.033 x Length) + 22.374 (r^2 = 0.3314).

Stomach contents

Most stomachs were relatively empty. Four stomachs were relatively full and these contained predominantly seagrass. The weights of stomach contents ranged from 13 to 95 g per fish. Unidentified digested material comprised 77% by volume of total material examined (ranging from 20–100% per stomach) and seagrass (*Posidonia sinuosa* and *P. australis*) comprised 19% by volume of total material examined (ranging from 0–80% per stomach). Invertebrates comprised about 1% of total material examined (ranging from 0–5% per stomach). Sand/rock and a feather were also present.

Discussion

To our knowledge, the only other published report of a mass stranding of *R. laevis* is also from the Albany region, where approximately 50 individuals washed ashore in 1928 (Gomon *et al.* 2008). Anecdotal reports from commercial fishers and others suggest that *R. laevis* strandings are relatively common along the south coast, with small numbers of fish being sighted here in most years (G. Kennedy & G. Jackman, pers. comm.). All past sightings of stranded fish along the south coast have occurred from April to June, coinciding with the maximum flow of the Leeuwin Current in this region. Unfortunately, none of the people we interviewed were able to recall the precise number of fish observed in any previous year or the particular year(s) in which sightings had occurred. However, all agreed that the number of *R. laevis* stranded along the south coast was substantially higher in 2008 than in any previous year.

In 2008, the strength of the Leeuwin Current was well above average. In addition, limited data suggest that the Leeuwin Current may have flowed unusually close to the shore in 2008. This current typically flows along the edge of the continental shelf and so beach temperatures usually reflect cooler shelf waters rather than the warmer waters of the Leeuwin Current. However, commercial fishers reported atypical ocean conditions at the time of the strandings, including above average beach water temperatures and the presence of various tropical species not commonly found near the shore in this region. The transport of a relatively high volume of warm water to the south coast by the Leeuwin Current, accompanied by a greater than average shoreward distribution of the current flow could explain the exceptional number of *R. laevis* occurring near the shore in 2008.

The above information suggests that *R. laevis* are transported to the south coast annually by the warm waters of the Leeuwin Current. *Ranzania laevis* is generally reported from waters >20 °C (Robinson, 1975; Sokolovskaya & Sokolovskiy 1975; Castro & Ramos 2002; Wan & Zhang 2005). The surface temperature of the Leeuwin Current as it flows along the south-western Australian continental shelf in autumn is usually >20 °C, which is typically 2–4 °C warmer than ambient surface temperatures (Rochford 1984; Smith *et al.* 1991; Cresswell & Peterson 1993). Hence, the surface waters of the Leeuwin Current are generally at a suitable temperature for *R. laevis* and could host individuals of this species whilst advecting them to the south coast.

A total of nearly 700 fish were observed on/near beaches in April–June 2008. It is likely that additional fish occurred at remote locations and were not observed. Therefore, the actual number of fish in the region at this time was almost certainly higher than that observed. Individuals and small schools were simultaneously observed in different locations that extended along 100's km of coastline, suggesting that these fish were actually part of a very large, dispersed aggregation of fish present along the south coast.

The advection of large numbers of *R. laevis* within a warm water mass has also been reported in the central east Atlantic. In May–July 2002, a very large aggregation of *R. laevis* comprised of solitary fish and schools of >500 fish was observed off the Canary Islands (latitude 28° N). Their appearance coincided with the arrival of a warm water mass (21–23° C), which was 3–5° C warmer than the surrounding surface waters (Castro & Ramos 2002).

The large aggregations of *R. laevis* along the south coast of Western Australia and off the Canary Islands were not associated with spawning. The Canary Islands aggregation was comprised of fish 20–25 cm in total length, which were probably juveniles (see below). The Western Australian aggregation probably consisted entirely of adults, but they were not in spawning condition. Spawning and gonad development in *R. laevis* has not been described although Jardas & Knezevic (1983) observed a single male with a "fully ripe testes" (probably equivalent to macroscopic stage 5) of *R. laevis* that comprised 9.13% of total body weight and occupied a major part of the abdomen. In contrast, most fish sampled in May 2008 possessed gonads that weighed between 0.53 and 1.37% of whole body weight. In addition, the presence of cortical alveoli oocytes with lipid droplets as the most advanced oocyte stage within all specimens microscopically analysed indicated that these fish were not spawning at the time of sampling.

Limited evidence suggests that *R. laevis* attain maturity at a total length of approximately 300 mm. Published reports of "adults" describe fish that are >300mm, whereas individuals reported to be juveniles or of indeterminate sex are <300mm (Sherman 1961; Fitch 1969; Robinson 1975; Jardas & Knezevic 1983; Ebenezer & Joel 1984; Iddison 2002; this study). *Ranzania laevis* is reported to attain a maximum total length of 900 mm (Hutchins 2001). However, we were unable to find a published account of any particular fish above 700 mm. Hence, the fish observed off south-western Australia in 2008 apparently comprised the full size range typical for adults of this species (*i.e.* 300 to 700 mm).

In 2008, stranded fish had apparently not fed for at least a few days prior to stranding. The stomachs examined were relatively empty except for seagrass. Vegetation has been found previously in stomachs of *Ranzania laevis* although it is unclear whether ingestion of vegetation by this species is incidental (Fitch 1969). Seagrass is abundant in the Albany region and large accumulations of dead leaves were present in shallow waters and on the beaches where fish were stranded. It is possible that seagrass was ingested incidentally, either while attempting to prey on invertebrates or during the process of stranding.

Incidental ingestion of seagrass during stranding is possible, given the permanently open mouth. Conflicting reports exist about the mouth morphology of *R. laevis.* The mouth opening forms a wide oval (with long axis in the vertical plane) and some authors have reported that it closes to form a vertical slit (Fraser-Bruner 1951; Ebenezer & Joel 1984). However, others have reported that it is a relatively inflexible structure and remains permanently open (Fitch 1969; Robinson 1975; Iddison 2002). The living and dead specimens observed in south-western Australia in 2008 had open mouths that were apparently inflexible and permanently open.

Evidence from the small number of fish examined by pathologists was inconclusive in regards to nutritional status. These fish displayed symptoms consistent with hepatitic lipidosis seen in some fish species, where lipids are mobilised in the liver in response to starvation or nutritional imbalance (Wolf & Wolfe 2005). However, members of the related family Tetraodontidae typically have a high lipid content in their livers and so this condition may also be typical in Molidae (Ando *et al.* 1993; Hazra *et al.* 1998). Further sampling to determine 'normal' liver condition in *Ranzania* would be required.

The cause of strandings is not clear. Adult *R. laevis* are normally strong, agile swimmers that would be capable of avoiding stranding (Gomon *et al.* 2008). Adults have been observed swimming rapidly near the surface and even jumping completely out of the water (G. Kennedy & G. Jackman, pers. comm.). Burst speeds of 90–100 cm/second have been reported for juveniles (Robinson 1975). Also, the structure of the fins and musculature are suggestive of fast swimming (Raven 1939). However, the fish that stranded were evidently weak and disoriented. In 2008 and in earlier years, stranded fish were 'rescued' by members of the public and commercial fishers who returned them to deeper waters, but the fish quickly swam back to the beach and became stranded again. Some fish were observed to strand during very calm weather, and so storm surge was not a primary cause of strandings.

In summary, the reason for the mass strandings of *R. laevis* off south-western Australia in 2008 and in previous years remains unclear. However, it is likely that individuals of this species are transported to this region annually by the Leeuwin Current. This current mainly flows over the shelf break but eddies and meanders could occasionally transport fish towards the shore. Starvation due to low prey availability within the Current may weaken individuals en route to the south coast and contribute to strandings.

Given the apparent relationship between strandings and the Leeuwin Current, it is likely that strandings in this region will continue to occur periodically and could be forecast from oceanographic data. In 2008, a substantial amount of data, including specimens and numerous photographs, were provided by the public. In future, the length-height ratio could be a useful method of estimating fish length from photographs provided by the public.

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New records of *Galaxias truttaceus* (Galaxiidae) in the Kent River catchment, southwestern Australia

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Abstract

A number of incidental captures of the Trout Minnow, *Galaxias truttaceus*, have been made recently in the Kent River catchment in southwestern Western Australia. This freshwater galaxiid was thought previously to be confined to a number of river systems east of Albany in Western Australia. Between 2005 and 2009, four specimens of *G. truttaceus* were captured in the Kentdale Creek, which flows into the Kent River west of Denmark. All four specimens were large, presumably mature, individuals, with total lengths (TL) ≥140 mm, and the largest was 12 mm longer than the maximum size previously recorded in Western Australia. The Kentdale Creek dries completely between December and May, and hence it is likely that the fish either move into the adjacent stretch of the Kent River, which continues to flow even during the driest months, or they perish. A dedicated study is required to determine the size, distribution, biology, migration patterns and status of this newly discovered population of the Trout Minnow, Australia's only freshwater fish that is listed as critically endangered.

Keywords: Trout Minnow, Kentdale Creek, incidental captures, Owingup Swamp, southwestern Australia, landlocked or diadromous life style

Introduction

The Trout Minnow, *Galaxias truttaceus* Valenciennes, 1846, is a native freshwater fish that occurs in coastal drainages of Tasmania, southern Victoria, major islands of the Bass Strait and also a number of rivers in southwestern Western Australia (Allen *et al.* 2002). It is the rarest galaxiid in Western Australia, being restricted to small catchments of the Goodga and Kalgan Rivers east of Albany (Morgan 2003). The Western Australian population of *G. truttaceus* was originally described as a discrete taxon, *Galaxias truttaceus hesperius* Whitely, 1944, but was revised as a junior synonym by McDowall & Frankenburg (1981). In 2006, the Western Trout Minnow was listed as critically endangered under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), currently making it Australia's only critically endangered freshwater fish. The species (as *G. truttaceus hesperius*) is also listed under Schedule 1 of the Western Australian *Wildlife Conservation (Specially Protected Fauna) Notice 2010* (Fauna that is rare or is likely to become extinct).

Historically, the Trout Minnow had been recorded from the catchments of the King and Kalgan Rivers (McDowall & Frankenburg 1981; Allen 1982; Morgan *et al.* 1998). Sampling between 1994 and 1996 (Morgan *et al.* 1998), and more recent surveys by the Centre for Fish and Fisheries Research (CFFR) at Murdoch University (Morgan 2003) did not record this species in either the King or Kalgan Rivers. From these surveys it was thought that the range of *G. truttaceus* was restricted to a 4 km stretch of the Goodga River and approximately 2 km of stream in the Angove River (TSSC 2006). The biology of the Goodga River/Moates Lake population of *G. truttaceus* is described in detail in Morgan (2003). These previous surveys (Morgan *et al.* 1998; Morgan 2003; TSSC 2006) included a number of sampling sites on the Kent and Styx Rivers and their tributaries, to the west of Denmark.

Apart from the cluster of known occurrences in the catchments of the Goodga River/Moates Lake and Angove River, the only other confirmed record (from a search of the NatureMap database; Department of Environment and Conservation 2010) for the Trout Minnow is a single specimen collected from the Owingup Creek (on Board Road, approximately 23 km west of Denmark) in 1990 (Dr Sue Morrison, Western Australian Museum, pers. comm., January 2010). The longitude for this record was incorrectly entered into the Western Australian Museum database, and this error was consequently carried through to Morgan *et al.* (1998: Table 10, page 88). The Owingup Creek is not part of the Kent River, as it flows into Owingup Swamp.

This paper describes the discovery of a population of the Trout Minnow in the Kentdale Creek, which flows into the Kent River. At present, this population is known only from four separate specimens, captured between October 2005 and October 2009.

Methods

The Kentdale Creek is situated approximately 27 km west of Denmark on the southern coast of Western Australia (Figure 1), and flows westward through farmland, remnant karri forest and tree plantations into the Kent River.

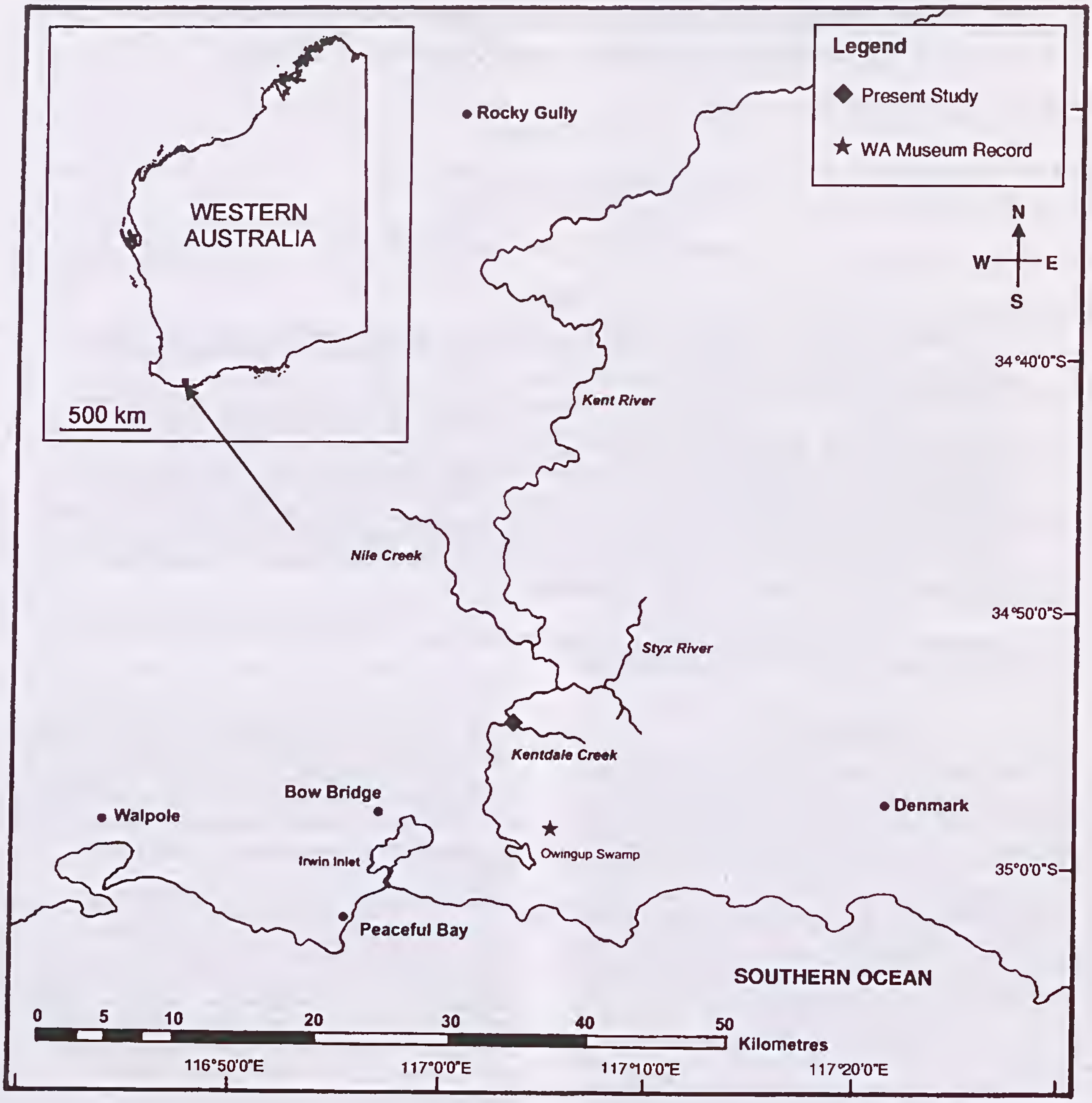

Figure 1. Location of incidental captures of *G. truttaceus* in the Kent River catchment, and historical record for this species in the Owingup Creek.

The catchment area of this system is approximately 7.5 km^2, and all of this area is on private property or unallocated crown land. Only the lower 250 m length of the creek was sampled in this study, between where it flows through a culvert under Parker Road and the junction of the creek with the Kent River. All of this lower extent of the creek, which is approximately 1.5–2.0 m wide but includes a number of larger pools, is located on a 1.35 ha forested block owned by the author. From the study area, the Kent River flows south and drains into Owingup Swamp (Figure 1).

Sampling in the creek was restricted to a number of random episodes covering a few days in October and November each year, when the creek was at its maximum height. Additionally, some sampling was conducted sporadically in the adjacent stretch of the Kent River, to a distance of approximately 200 m upstream and downstream of the junction with the creek. Fish were captured using either a hook and line, baited with worms, or a collapsible fish trap (600 x 400 x 250 mm rectangular volume enclosed in a 10 mm nylon mesh, with two funnel entrances giving access to a baited pouch). The fish trap was baited with a variety of baits, including commercial marron food, and it was used in areas where the water depth was deeper than 400 mm. Neither capture method would have captured small fish—the sampling was biased to larger sized specimens and therefore these records represent only incidental captures. No repeated sampling was undertaken as part of this study.

Captured *G. truttaceus* specimens were photographed, and the total length (TL) of each specimen was measured prior to the fish being either released alive back into the creek, or preserved for taxonomic identification and genetic studies at the CFFR.

Results

Over the period October 2005 to October 2009, a total of four specimens of *G. truttaceus* were captured from the lower stretch of the Kentdale Creek (Table 1).

Specimen A, captured in October 2005, was not photographed or measured prior to being returned alive to the creek. Based on a preliminary identification that this fish could be *G. truttaceus*, further opportunistic sampling in Kentdale Creek was undertaken in early November 2005, and Specimens B and C were taken during a two day sampling episode. After photographs and TL measurements were made, both specimens were euthanised by chilling. Specimen B was preserved in methylated spirits and Specimen C was frozen, prior to being transported to Perth and passed on to Dr David Morgan at the CFFR. Specimen D, captured in early October 2009, was photographed and measured before being returned alive to the creek. Photographs of Specimens B, C and D are provided in Figure 2.

Table 1

Capture dates, capture method and TL measurements for the four specimens of *G. truttaceus* taken from the Kentdale Creek.

Specimen	Capture date	Capture method	Size (TL mm)
Specimen A	05/10/2005	Hook and line	140 (estimated)
Specimen B	11/11/2005	Hook and line	140
Specimen C	12/11/2005	Baited fish trap	165
Specimen D	08/10/2009	Baited fish trap	180

All four specimens had TLs ≥140 mm (Table 1; Figure 2) and the two largest specimens (C and D) were both captured using the fish trap in the same large pool on the creek. It is possible that Specimens A and B were the same individual, as both were captured within a few weeks of each other at the same location on the creek. Despite sporadic sampling on other occasions during late winter/early spring between 2005 and 2009, in both Kentdale Creek and adjoining stretch of the Kent River, no other specimens of *G. truttaceus* were captured. Hence, at present this population is known only from these four incidental captures.

Genetic samples and measurements have been taken from Specimens B and C by researchers at the CFFR.

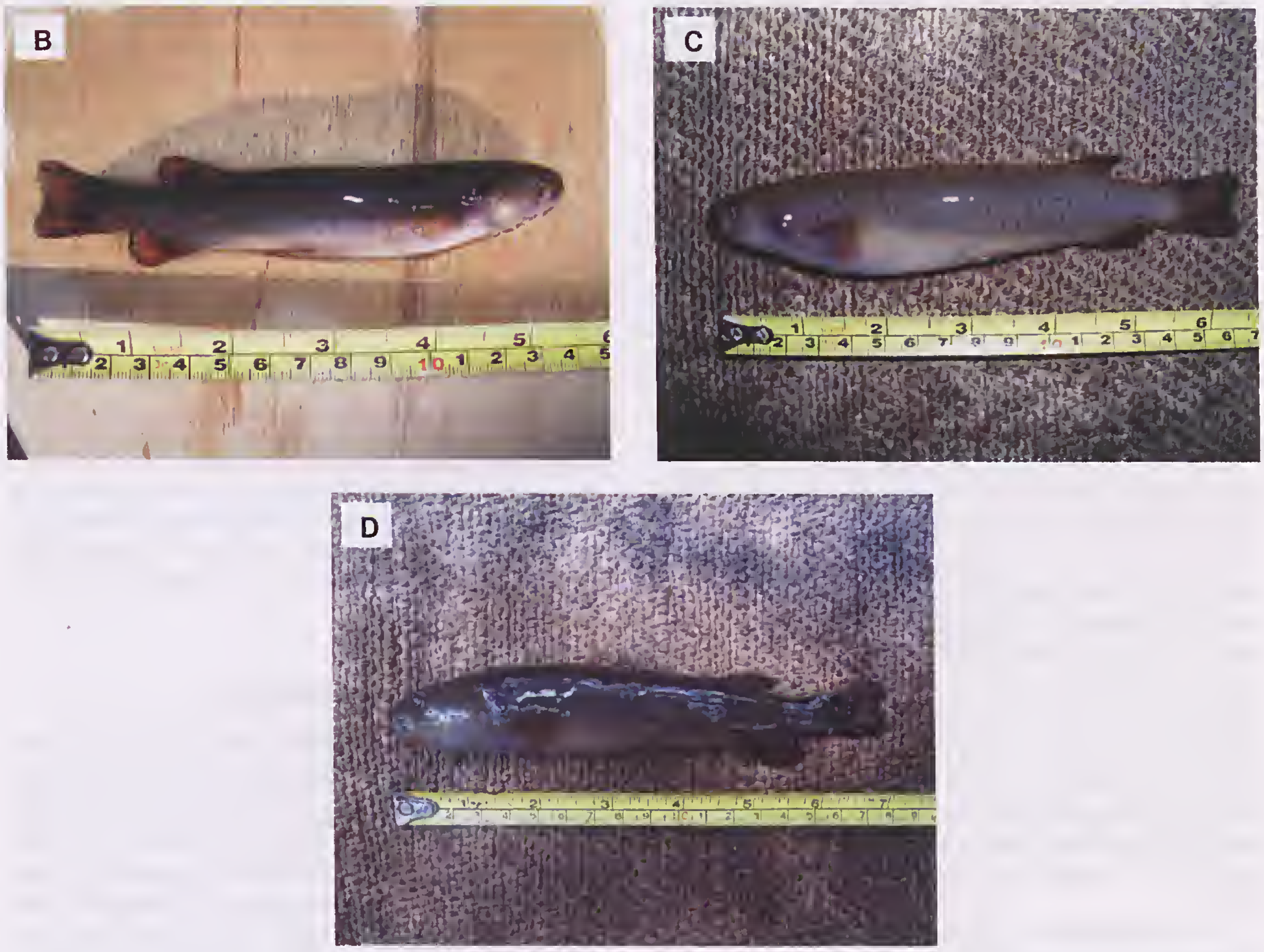

Figure 2. Photographs of *G. truttaceus* specimens captured in the Kentdale Creek in November 2005 (B and C) and October 2009 (D).

Both specimens are currently stored in 100% ethanol and will eventually be lodged in the Western Australian Museum fish collection (Dr David Morgan, CFFR, pers. comm., August 2009).

Large numbers of another native galaxiid, the Western Minnow (*Galaxias occidentalis* Ogilby, 1899), were captured in both the Kentdale Creek and the adjoining stretch of the Kent River using both sampling methods. Some specimens of Western Pygmy Perch (*Edelia vittata* Castelnau, 1873) were also taken using the fish trap. *E. vittata* and *G. occidentalis* are the most common and widespread of the freshwater fishes endemic to southwestern Western Australia (Morgan *et al.* 1998; Allen *et al.* 2002).

Discussion

The Kent River population of the Trout Minnow occurs at a location approximately 94 km further west along the south coast from the Goodga River/Moates Lake location, which represents a significant range extension for this rare, and critically endangered, galaxiid. Previously, the species was thought to be restricted to two small populations occurring along a 4 km stretch of the Goodga River and approximately 2 km of stream in the Angove River, with an estimated extent of occurrence of just 0.012 km^2 (TSSC 2006). The occurrence of *G. truttaceus* in vicinity of the Kent River area (in the Owingup Creek) has not been realised previously, as a result of an incorrect record for the specimen in the Western Australian Museum database.

The Kentdale Creek dries completely during the period December to May and usually there are no remnant habitats along the length of the creek during this period. Therefore, any fish utilising it as a habitat must either move downstream into the adjoining stretch of the Kent River, which continues to flow even during the driest months, or perish. Therefore, it is likely that the population uses the Kent River as its main refuge, and opportunistically colonises seasonal creeks. The extent of the population of the Trout Minnow in the Kent River is currently unknown.

All four specimens captured were large, adult fish with TL $\geq$140 mm (Table 1; Figure 2). These were presumably mature individuals, as Morgan (2003) reported a length at first maturity of 73.4 mm for females and 59.4 mm for males for the Goodga River/Moates Lake population of *G. truttaceus*. The largest individuals recorded from the Goodga River measured 151 mm TL, while the largest from the Angove River measured 168 mm TL (Morgan & Beatty 2005, 2006). Hence, the largest of the specimens captured in the Kentdale Creek (Specimen D, 180 mm TL) was 12 mm longer than the maximum size previously recorded in Western Australia. It has to be recognised that the records from the Kentdale Creek are incidental captures only, and did not result from any intensive sampling effort. As the capture methods used would tend to select larger specimens, no conclusions concerning population structure can be made.

It is possible that spawning in the Kent River population occurs in mid to late autumn, as it does in the Goodga and Angove Rivers (Morgan 2003). The population could have either a landlocked or diadromous life style. If the population is landlocked it is likely that the adults move upstream to spawn and then the larvae are transported downstream to Owingup Swamp, where they feed for the next 2–3 months before the juveniles migrate back up into the Kent River and its tributaries. If the population is diadromous, the adults would migrate downstream to the Irwin Inlet to spawn, with larvae spending some time feeding in the open sea.

Owingup Swamp drains into the Irwin Inlet via swamplands and a channel connecting the two water bodies. The Irwin Inlet is usually open to the Southern Ocean every winter, with the natural and, in some years, artificial removal of the sandbar at its mouth (Hodgkin & Clark 1988; Bancroft *et al.* 1997). Therefore, the Kent River population of *G. truttaceus* could be either landlocked, as is the case with the Goodga River population (Morgan 2003), or diadromous, which is the life style of several populations in southeastern Australia (McDowall & Frankenburg 1981; Humphries 1989, 1990). This would have to be determined by further studies.

The only other confirmed record for *G. truttaceus* in the Kent River area is a specimen captured from the Owingup Creek, which also drains into Owingup Swamp (Figure 1). Despite several surveys (Jaensch 1992; Morgan *et al.* 1998) no records of juvenile or adult Trout Minnow exist for Owingup Swamp, which has been classified as the most important wetland across the south coast of Western Australia in terms of species richness, with eight native fish species recorded (Jaensch 1992). Owingup Swamp, which is contained wholly within the Owingup Nature Reserve, may be critical for the survival of the Kent River Trout Minnow population, as it may (if the population has a landlocked life style) provide the lacustrine environment that functions as a nursery and feeding area for larval fish before they migrate back up the Kent River in the spring.

Significant threats to the Kent River population of *G. truttaceus* probably include declining water quality (as a result of land clearing and increasing salinity and nutrient levels in the Upper Kent River Catchment), loss of riparian vegetation, and stock access along the banks of its tributaries. The use of biological poisons on blue gum plantations in the catchment, the introduction of exotic fish species, disease and parasitic infections, and the construction of dams and weirs (which introduce an impassable barrier to upstream migration of juvenile fish) could represent additional threats to this population (Morgan *et al.* 1998; Morgan 2003; Gillespie 2006; TSSC 2006).

Conclusion

At present, the Kent River population of the Trout Minnow is known only from four incidental captures along a short length of the Kentdale Creek between 2005 and 2009. A dedicated study is required to determine the size, distribution, biology, migration patterns and status of the Kent River population of Australia's only freshwater fish that is listed as critically endangered. This research effort, which should be consistent with the objectives and actions of the Recovery Plan for *G. truttaceus hesperius* (Department of Environment and

Conservation 2008), will provide a better understanding of potential threats to the population and ensure its ongoing protection and conservation.

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A note on calcite precipitates as encrustations around sea rush roots and rhizomes and as laminae in high tidal zones of western Leschenault Inlet estuary

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Abstract

Precipitates of calcite occur in the high tidal zone of the western shore of the Leschenault Inlet estuary as encrustations around roots and rhizomes of the sea rush *Juncus kraussii*, as permineralised plant and fungal material, as laminae transgressing the encrustations, and as laminae in mud and muddy sand. The calcite forms crusts around the roots and rhizomes, as bladed crystals 0.5–1.0 mm in length, preserving the roots/rhizomes as casts, as thin horizontal sheets and micro-lenses within the sediment, and cementing the sediment. Mineralogically the precipitates are calcite and some Mg-calcite. Crystallographically, the crusts are layered, with crystal growth hiatuses marked by dissolution of calcite and by Fe-oxide and Mn-oxide precipitates. The hydrological and hydrochemical setting of the calcite precipitates is where freshwater seepage from the quartz and calcareous sand of the dune barrier delivers calcium carbonate-charged waters to the high-tidal flat. These types of encrustations and calcite laminae are absent from the eastern quartz sand shore of the estuary.

Keywords: sea rush rhizoliths, calcite encrustations, calcite laminae, permineralisation, tidal zone, estuary, Leschenault Inlet estuary

Introduction

The formation of rhizoliths, or rhizocretions, in paludal, lacustrine, riverine, estuarine, high tidal flat, and palaeosol environments by mineralisation, biomineralisation, permineralisation, or concretion of plant roots, including their calcitisation, has been documented for ancient sequences from Carboniferous to the Pleistocene (Cohen 1982; Mount & Chen 1984; Wright 1986; Liutkus *et al.* 2005). Mostly, the calcitisation or other forms of mineralisation and biomineralisation occur in ancient sequences with inferences about environment of deposition determined from stratigraphic and palaeoenvironmental reconstructions. Modern analogues of such shoreline mineralisation and rhizolithification of vegetation, and the calcreting of plant roots in soils are generally lacking. Exceptions are the mineralisation of plant roots colonising the margins of Lake Bogoria, Kenya, forming *in situ* rhizoliths, composed of various minerals (Owen *et al.* 2007), diagnetic clay concretions around plant roots on the tidal shore of the Saint Lawrence River (Rousseau 1934), and tuart-tree-induced precipitation of calcrete around roots and root hairs in the Leschenault Peninsula (Semeniuk & Meagher 1981a). This contribution describes calcite precipitation in the high tidal zone of the western shore of Leschenault Inlet estuary that results in the calcitisation of the roots of fringing estuarine vegetation, and development of calcitic laminae within the muddy sediment, the former resulting in calcitised rhizoliths or rhizocretions.

The setting of the calcite encrustations and laminae precipitates

The Leschenault Peninsula, a linear dune barrier in south-western Australia, located in a subhumid climate with mean annual rainfall of 880 mm, and annual evaporation of 1300 mm, is composed of parabolic dunes in various stages of mobility and fixation (Semeniuk & Meagher 1981b). Leeward of the barrier is the Leschenault Inlet estuary, an elongate shore-parallel, shallow water estuarine lagoon, bordered to the east by a Pleistocene quartz sand ridge, and to the west by a Holocene quartz and calcareous sand dune barrier (Figure 1; and Semeniuk 2000). The estuarine lagoon is microtidal, wave dominated and wind current driven, and its waters are annually poikilohaline, with marine, brackish, freshwater, and hypersaline fields (Wurm & Semeniuk 2000). Leschenault Inlet estuary has complex western shores as parabolic dunes encroach into the aquatic environment producing varied stratigraphic/hydrologic situations. The shores support a variety of fringing vegetation linked to the geomorphology, stratigraphy, hydrology and hydrochemistry (Cresswell 2000; Pen *et al.* 2000). Along the tidal zone there is a high-tidal platform, underlain by mud or muddy sand (Semeniuk 2000), inhabited by the sea rush *Juncus kraussii* at the interface between dune barrier and high-tidal platform. The rush forms closed cover, and its roots and rhizomes produce a mat up to 10 cm thick, comprising intertwining roots, root hairs, and rhizomes. Estuarine waters inundate the high-tidal flat surface in winter, and the groundwater table is 5–10 cm below the surface in summer. This high-tidal zone is the location of the calcitic encrustations and laminae, which occur in isolated

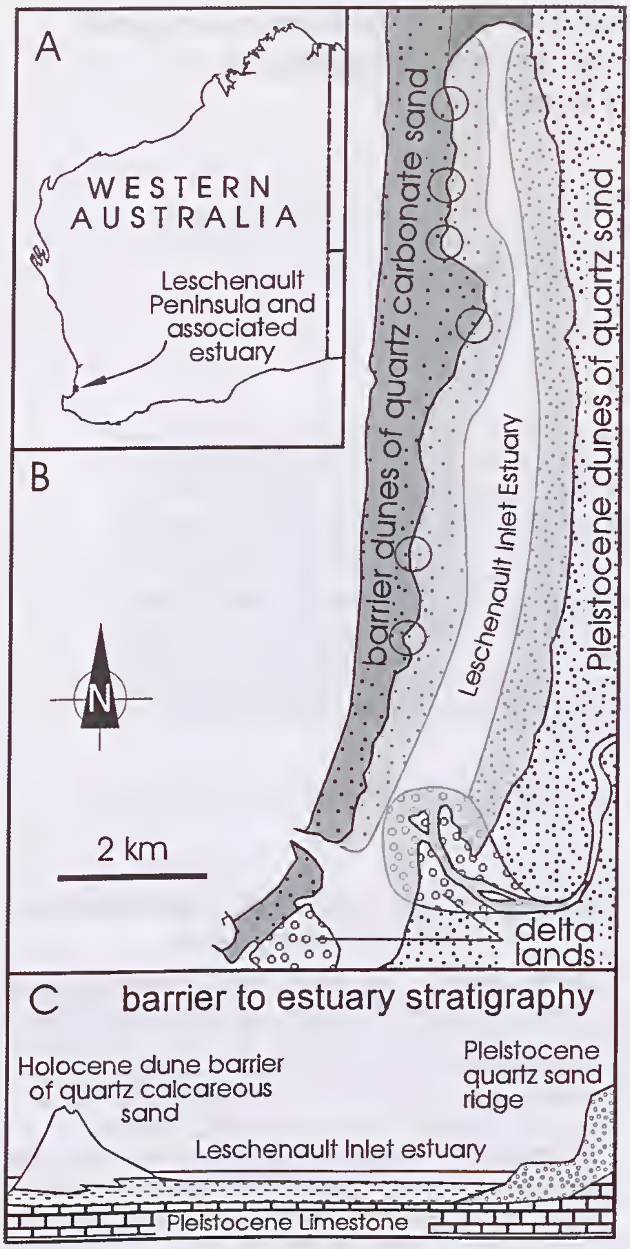

A
WESTERN AUSTRALIA
Leschenault Peninsula and associated estuary
B
N
2 km
barrier dunes of quartz carbonate sand
Leschenault Inlet Estuary
Pleistocene dunes of quartz sand
delta lands
C barrier to estuary stratigraphy
Holocene dune barrier of quartz calcareous sand
Pleistocene quartz sand ridge
Leschenault Inlet estuary
Pleistocene Limestone

Figure 1. A. Location of Leschenault Peninsula and its associated estuarine lagoon. B. The setting of the Leschenault Inlet estuary between the Holocene dune barrier and the Pleistocene quartz ridge. Circled areas show sampling sites for specimens described in this paper. C. Stratigraphic setting of the Leschenault Inlet estuary (after Semeniuk 2000).

patches and lenses (rather than a continuous shore-parallel ribbon). The calcite precipitations are absent from the quartz sand eastern shore of the estuary.

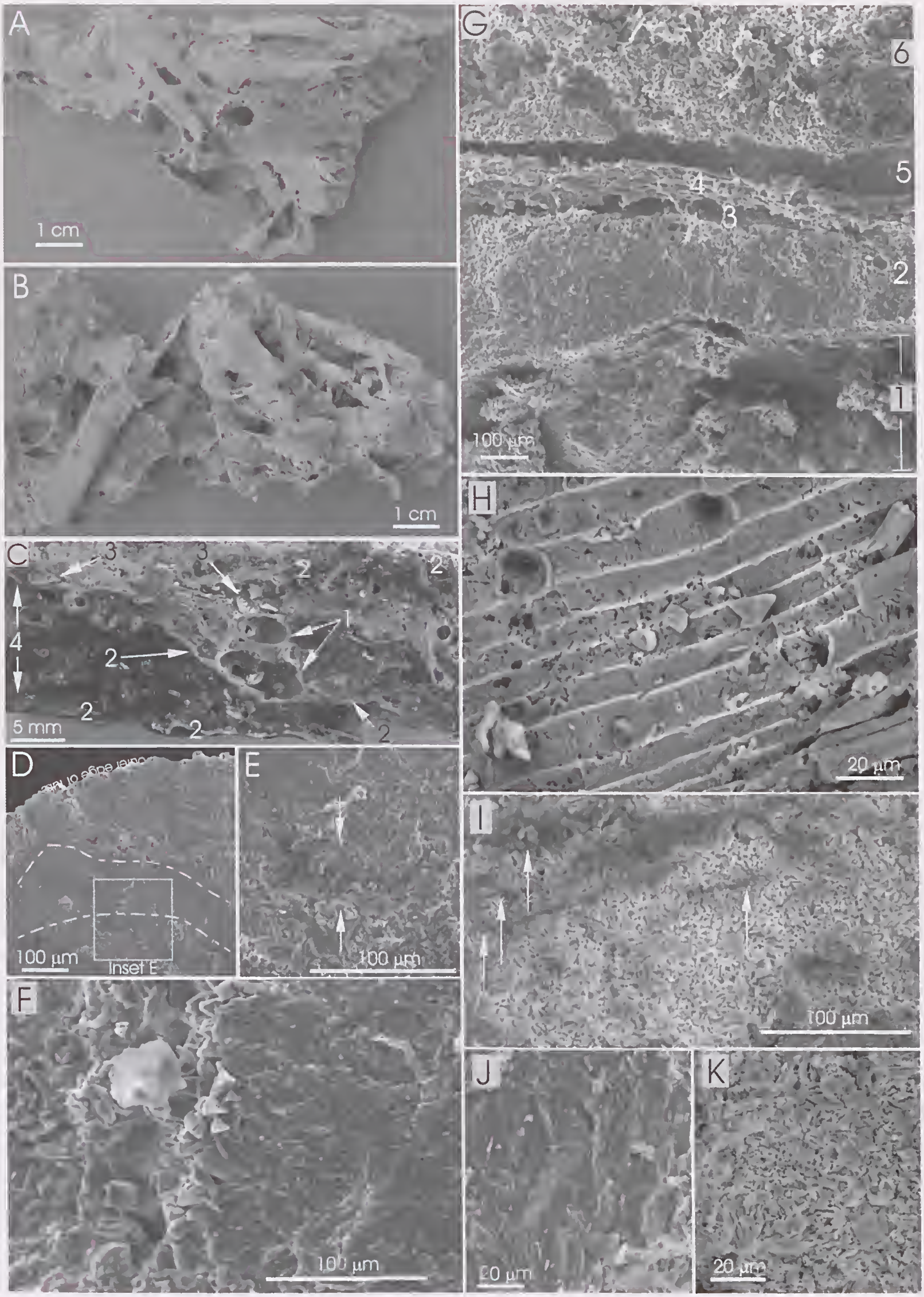
Within the dune barrier there is a prism of freshwater, bordered to sea and to the estuary by saline waters. Freshwater discharging seawards results in beach rock (Semeniuk & Searle 1987). Freshwater discharging to the estuary results in variable ecological responses (Cresswell 2000; Pen *et al.* 2000; Semeniuk *et al.* 2000), and in precipitation of calcite in the muddy platform. The salinity of the groundwaters under the dune barrier near its margin with the estuary is 500–1030 ppm, and where it interfaces with the high-tidal platform it is 880–1050 ppm (Semeniuk & Meagher 1981; Cresswell 2000; Semeniuk *et al.* 2000). Freshwater seepage from the dunes into the high-tidal mud platform environment has Ca concentrations of 118–124 ppm (Semeniuk & Meagher 1981a). The salinity of the groundwaters under the rushes varies from 630–8300 ppm (Cresswell 2000; Semeniuk *et al.* 2000).

The calcite encrustations and laminae

The precipitates of calcite occur in five forms (Figure 2): 1. as encrustations on *Juncus kraussii* roots/rhizomes; 2. as permineralised plant matter; 3. as permineralised fungal threads; 4. as thin discontinuous laminae associated with the encrustations, or embedded in mud and muddy sand and displacive of sediment; and 5. interstitial to sediment. The encrustations are localised mainly on *in situ* rush roots/rhizomes (Figure 2A and 2B). After the roots/rhizomes have decayed, the precipitation structures are tubular. Since *Juncus kraussii* roots and rhizomes form an inter-twining mat, the encrustations also form an intertwining meandering and branching mass of tubes, as a sheet 1–3 cm thick, commensurate with the thickness of the mat. The geometry and intertwining of the roots and rhizomes are faithfully recorded by the encrustations. Sediment formerly between the roots and rhizomes is incorporated as inclusions in the encrustations, and also is cemented interstitially by precipitate peripheral to the encrustation. The encrustations around plant roots are 400–1500 μm thick with concentric layers, 50–200 μm thick, of varied crystal mosaics and structures (Figure 2D, 2E, 2F, 2G, 2H, 2J): radially oriented bladed crystals (the dominant type), randomly oriented discoid crystals, or permineralised fungal hyphae (2.5–7 μm in diameter and > 200 μm long)

Figure 2. A and B. Horizontal views of exhumed calcitised *in situ* roots and lower stem of *Juncus kraussii*. C. Polished cross section of (a resin-impregnated slab) of calcitised roots of *Juncus kraussii* showing (1) tubes where former roots were encrusted by calcite, (2) laminae of calcite, (3) molluscs encased in the calcitic structure, and (4) calcite-cemented sediment (= dark zone). D–I. SEM images of the samples illustrated in A, B and C. D. SEM image of cross section of the encrustation around a root (the curved outer edge of the tubular encrustation is noted); the dashed lines separate the general trend of three concentric bands within the encrustation. E. Close-up of (D) showing laminoid cavity (positioned between the arrows) embedded in a mosaic of interlocking calcite crystals, with free terminations of calcite projecting into the cavity. F. Close-up of tightly packed bladed calcite, with free terminations projecting into a laminoid cavity. G. SEM image of cross section of the wall of an encrustation around a root. The encrustation shows 6 layers: (1) in the interior of the encrustation, a tangled mass of permineralised fungal hyphae and scattered discoid calcite crystals; (2) a zone of tightly packed bladed calcite crystals oriented long axes radial to the tube; (3) permineralised plant matter, with cell structure; (4) permineralised fungal hyphae; 5. permineralised plant matter showing cell structure; and (6) towards the exterior of the encrustation, randomly oriented discoid calcite crystals. H. Close-up permineralised plant matter showing its cell structure. EDS of five sites in this field of view shows the Mg content of the calcite to vary from 1.6 to 5.5. Mol %. I. SEM image of cross section of a calcite lamina comprised of tightly packed randomly oriented discoid calcite crystals (see K below); four discontinuities in the lamina marking internal (finer laminar) structures are arrowed. J. SEM image showing close-up of a tightly-packed bladed calcite crystal mosaic (oriented vertically) from a root encrustation. K. SEM image showing close-up of a mosaic of tightly-packed randomly oriented discoid calcite crystals from a lamina.

▶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enveloping the *Juncus* roots, or permineralised roots that show plant cell structure Figure 2G, 2H). The encrustations have local lensoid to laminar cavities parallel to the concentric layering, 15–40 μm to 100 μm high, and ~ 150–2000 μm in length, marked by calcite crystals with free terminations (Figure 2E, 2F). Calcite precipitates also form thin laminae interlayered with the root/rhizome encrustations (Figure 2C), or isolated micro-lenses (or platelets) in the mud and muddy sand, and occasionally small nodules. The calcitic laminae and micro-lenses generally form sheets 1.0–0.05 mm thick, that are flat to undulating, occurring at different levels in the upper 30 cm of the sediment profile (Figure 2C). The isolated platelets are 1.0–0.05 mm thick, as small discontinuous bodies (generally varying from 2 cm across to as small as 1 cm across, locally forming larger platelets 5 cm across and 2 mm thick). The more continuous laminae are 20 cm long, and transgress primary sedimentary structures. The small nodules are 2–5 mm in size.

Thin sections and Scanning Electron Microscopy (SEM) show the encrustations, permineralisations, and laminae to be comprised of one or more of five types of calcite aggregates: 1. even textured, epitaxial micro dogtooth spar (with free terminations where they project into cavities or form the outer crust) that predominantly comprise encrustations; they are generally 5–10 μm long, but in some mosaics 10–40 μm long (Figure 2E, 2F); 2. even textured, interlocking, long-bladed spar that comprises encrustations and laminae; they are generally 10–20 μm long, but up to 50 μm long (Figure 2F, 2J); 3. even textured, interlocking randomly oriented discoid crystals (flattened scalenohedra), 5–15 μm in size to 20–40 μm in size, that comprise the laminae and zones in root encrustations (Figure 2K); 4. very fine grained equant crystal aggregates (< 1 μm in grainsize) that comprise the permineralisations of fungal hyphae (zone 4 in Figure 2G), or of plant matter (zones 3 and 5 in Figure 2G), the former occurring as thread-like filaments, the latter showing plant cell structure; and 5. a tangled mass of permineralised fungal hyphae, with embedded scattered discoid calcite crystals (zone 1 in Figure 2G). Within epitaxial crusts in the encrustations and laminae, crystal growth hiatuses are marked by dissolution, and by Fe-oxide and some Mn-oxide precipitates (Figure 2I). For each of the SEM images shown in Figure 2, textural changes, crystal aggregate zones, and permineralisations were analysed for content of Ca, Mg, Fe, and Mn using Energy Dispersive Spectroscopy (EDS). SEM topography and EDS results show that while (low Mg) calcite dominates the mineralogy, Mg substitution in the calcite varies from 1.6 to 10 Mol % Mg, thus Mg-calcite (4–10 Mol % Mg) is also present. X-Ray Diffractometry shows the precipitates to be mainly calcite. Locally, a very thin layer of fine grained dolomite, < 1 μm thick, and composed of crystals < 1 μm in size, coats the encrustations, permineralised materials, and the laminae, but dolomite is a very minor component of the mineralogy.

Discussion

The precipitates of calcite around roots and rhizomes of the rush *Juncus kraussii*, as permineralisations, and as laminae and micro-lenses, in the shore environment of the Leschenault Inlet estuary, provide a Holocene model for similar rhizoliths, permineralisations, and calcitic laminae micro-lenses recorded in the geological record. The hydrogeological setting of the barrier and the precipitates along the shores of the Leschenault Inlet estuary points to Ca^{2+} and CO_3^{2-} charged groundwater discharging as seepages from the quartz and calcareous sand parabolic dunes. In effect, the high-tidal precipitates along the estuarine shore are the hydrogeological and diagenetic equivalent of beach rock that occurs on the oceanic western shore of the dune barrier. The internal features of the precipitates indicate that there is precipitation and dissolution of calcite crystals, with periods of dissolution marked by precipitation of Fe-oxides and Mn-oxides. The rushes and fungi appear to have biomediated the precipitation of the calcite and the metal oxides. The laminae and platelets of calcite, transgressing primary sedimentary structures, indicate that precipitation is related to a groundwater table, and the various levels of the calcitic laminae in the shallow stratigraphic profile record the varying positions of the summer water table.

Rhizoliths recorded from modern lake shores and tidal rivers banks elsewhere globally are different from those of Leschenault Inlet estuary in terms of mineralogy and formative processes. The mineralisation of plant roots around the margins of Lake Bogoria in Kenya forms root casts, moulds, tubules, rhizocretions, and permineralised root systems, and involves a variety of minerals, *viz.*, opaline silica, calcite, zeolites, and fluorite and a variety of hydrochemical processes (Owen *et al.* 2007). Silica precipitation and permineralisation of plant tissues are associated with hot-spring fluids, concentrated by evapotranspiration and capillary evaporation. Calcite precipitation is the result of evaporative concentration, evapotranspiration, and/or CO_2 degassing of Ca-bearing runoff water that infiltrated the sediment, or mixing of runoff with saline, alkaline groundwater. Fluorite precipitation results from mixing of hot-spring and meteoric waters or hot-spring fluids coming into contact with pre-existing calcite. Zeolite precipitation is the result of prolonged periods of aridity when capillary rise and evaporative pumping brings saline, alkaline waters into contact with detrital silicate minerals around roots (Owen *et al.* 2007). On the tidal shore of the Saint Lawrence River, the diagnetic induration of clays as casts around *Juncus* and *Scirpus* plant roots (Rousseau 1934) involves Fe-sulphide alteration of clay around decaying plant roots, and its later oxidation to Fe-oxide and hence induration of a clay envelope around the roots. These provide contrasting mineralogy and mineralising mechanisms for development of rhizoliths to that in the Leschenault Inlet estuary.

To date, calcitisation and permineralisation of rush roots and fungal filaments have not been recorded elsewhere in peripheral vegetation of estuaries in south-western Australia because parabolic dunes and quartz and calcareous sand dunes are not major shore types of these estuaries. Though they have local occurrences of muddy high-tidal platforms, the shores of the Swan River estuary and the Peel-Harvey estuary are bordered mainly by quartz sand terrains (*i.e.*, they are carbonate depauperate), and barrier dunes of the southern coast estuaries are generally also carbonate depauperate. Similarly, calcitised and permineralised rush roots have not even been found on the quartz-dominated eastern shore of the Leschenault Inlet estuary. A carbonate-bearing dune upland, with parabolic dune incursions into the estuarine aquatic zone (and its attendant groundwater incursions and freshwater seepages) would seem to be requisites for the development of calcitic precipitates in the high-tidal zone of an estuary. As such, with the parabolic dune encroaching into the estuary and providing corridors of calcium carbonate charged freshwater seepages (Cresswell 2000), the western shore of the Leschenault Inlet estuary is hydrogeologically and hydrochemically distinct in Western Australia. This situation also renders the western shore of the Leschenault Inlet estuary diagenetically distinctive.

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